Chapter 4

Impacts of Human Disturbances on the Functions of Wetlands

4.1 Reader's Guide to This Chapter

Chapter 4 integrates the concepts discussed in Chapter 2 and Chapter 3. Chapter 2 described the functions performed by wetlands and the environmental factors that control these functions. Chapter 3 discussed the major disturbances caused by different human activities and uses of the land. This chapter continues by summarizing how each of the disturbances ultimately leads to impacts on wetland functions.

4.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 4.2, Introduction and Background on the Scale of Impacts to Wetland Functions describes how disturbances that impact functions in wetlands can occur either within the wetland itself or in the surrounding landscape. While the literature generally does not distinguish the scale of the disturbance when assessing impacts on wetland functions, there are some disturbances at the site scale that can remove all or most functions of the wetland (such as changing the physical structure of the wetland through filling).

Following this introduction, the chapter continues by describing how the major types of disturbances resulting from human activities affect wetland functions. As discussed in Chapter 3, different land uses may create the same type of disturbance (for example, both agriculture and urbanization may cause sedimentation). Therefore, each of the remaining sections of this chapter focuses on the different types of disturbances, without division by land use type, and their impact on each wetland function, as follows:

Section 4.3, Impacts of Changing the Physical Structure within a Wetland

Section 4.4, Impacts of Changing the Amount of Water in Wetlands

Section 4.5, Impacts of Changing the Fluctuation of Water Levels within a Wetland

Section 4.6, Impacts of Changing the Amounts of Sediment

Section 4.7, Impacts of Increasing the Amount of Nutrients

Section 4.8, Impacts of Increasing the Amount of Toxic Contaminants

Section 4.9, Impacts of Changing Acidity

Section 4.10, Impacts of Increasing the Concentrations of Salt

Section 4.11, Impacts of Decreasing the Connection between Habitats

Section 4.12, Impacts of Other Human Disturbances

Within each section, the impact of each disturbance is summarized in terms of the following wetland functions:

- Impacts on hydrologic functions
- Impacts on functions that improve water quality
- Impacts on plants
- Impacts on habitat for invertebrates
- Impacts on habitat for amphibians and reptiles
- Impacts on habitat for fish
- Impacts on habitat for birds (wetland-dependent species and wetland users)
- Impacts on habitat for mammals (wetland-dependent species and wetland users)

Section 4.13, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

4.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the key points resulting from the literature on that topic in a bullet list format. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 4.3.9
- Section 4.4.9
- Section 4.5.9
- Section 4.6.9
- Section 4.7.9
- Section 4.8.9

- Section 4.9.9
- Section 4.10.9
- Section 4.11.9
- Section 4.12.6

In addition, Section 4.13 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

4.1.3 Data Sources and Data Gaps

Data on some of the subjects related to the impacts of human disturbances on wetland functions are abundant for select areas in the state. For example, the Puget Sound Wetlands and Stormwater Management Research Program (Azous and Horner 2001) has provided numerous studies on how changes in land uses in a watershed affect the physical, chemical, and biological processes in wetlands of the Puget Sound lowlands.

Similarly, studies on the effects of changes in wildlife habitat resulting from physical changes within wetlands and reduced connection between habitats have been performed in Washington for some species and some types of habitat changes. The impacts to other species are less well studied or have only been examined in other states or other countries. Literature from other locales is included for these topics when relevant.

This chapter contains text that was adapted from a review of current scientific literature on the impacts of human activities on wetlands and their functions undertaken by the U.S. Environmental Protection Agency (Adamus et al. 2001). This review represents a very detailed summary of the literature published between 1990 and 2000 regarding wetlands across the United States. Portions of the review that were considered relevant to wetlands in Washington State were adapted for inclusion in this chapter, with permission from Dr. Adamus. The sections of this chapter that incorporate text adapted from the Adamus et al. (2001) review include:

- Section 4.3, Impacts of Changing the Physical Structure within a Wetland: Habitat for plants, invertebrates, reptiles and amphibians, fish, and mammals
- Section 4.4, Impacts of Changing the Amount of Water in Wetlands: Habitat for plants, invertebrates, reptiles and amphibians, fish, and birds
- Section 4.5, Impacts of Changing the Fluctuations of Water Levels within a Wetland: Habitat for invertebrates
- Section 4.6, Impacts of Changing the Amounts of Sediment: Habitat for plants and invertebrates
- Section 4.7, Impacts of Increasing the Amount of Nutrients: Habitat for plants, invertebrates, amphibians and reptiles, and birds

- Section 4.8, Impacts of Increasing the Amount of Toxic Contaminants: Habitat for plants, invertebrates, reptiles and amphibians, fish, and birds
- Section 4.9, Impacts of Changing the Acidity: Habitat for plants, invertebrates, reptiles and amphibians, and birds
- Section 4.10, Impacts of Increasing the Concentration of Salts: Habitat for invertebrates and birds
- Section 4.12, Impacts of Other Human Disturbances: Impacts to plant communities from altering soils, impacts of exotic invertebrates

The literature sources cited in the portions of the text that were adapted from the Adamus et al. (2001) report are included in the list of references at the end of Volume 1.

4.2 Introduction and Background on the Scale of Impacts to Wetland Functions

The disturbances that impact functions in wetlands can occur either within the wetland itself or in the surrounding landscape. Chapter 2 introduced the idea that the controls of wetland functions occur at both the "site scale" and the broader "landscape scale." As with the controls of wetland functions, disturbances caused by human activities can also occur at the same two scales (site and landscape).

For example, increased nutrients can flow into a wetland directly from an adjacent lawn or from animals grazing within the wetland (disturbance at the site scale). The nutrients could also originate from development or fertilizing fields somewhere higher in the contributing basin (disturbance at the landscape scale). As another example, the water levels in a wetland can be increased through the direct discharge of stormwater into the wetland (the site scale) or by adding impervious surface higher in the contributing basin (the landscape scale).

Much of the discussion in this chapter does not differentiate the scales at which the disturbance occurs. For example, the impacts on wetland functions resulting from excess nutrients or higher water levels can be expected to be the same whether they are delivered directly to the wetland or come from a distant source in the contributing basin. The literature does not usually differentiate between scales when discussing the impacts on wetland functions.

However, an alteration to the physical structure of the wetland itself is a type of disturbance that occurs only at the site scale. Filling, removing vegetation, tilling, or grazing within a wetland has a direct impact on the functions at that site. The most extreme impact to a wetland is the complete removal of all the factors that contribute to the existence of the wetland. Thus, filling a wetland or draining all the water eliminates all of the wetland functions because the wetland no longer exists.

4.3 Impacts of Changing the Physical Structure within a Wetland

Disturbances that directly change the structure of wetlands can be so severe that the wetland is destroyed. Filling or draining a wetland can so alter the water regime that the land can no longer support the wetland vegetation and maintain hydric soils. If a wetland is lost, most if not all of its wetland functions are also lost. Dahl (1990) estimated that 31% of the wetlands in Washington State had been lost prior to the 1980s as a result of filling or draining to the extent there is no longer enough water to maintain areas as wetland.

There are, however, some human alterations of the structure in wetlands that do not result in the complete loss of functions, including:

- Human removal of vegetation (for example, logging, mowing, or application of herbicides)
- Animal grazing
- Alteration of the soil through tilling or compaction
- Partial draining

This section describes what the literature says about how these alterations impact wetland functions. The impacts of grazing and removal of vegetation are better understood than those of alterations to the soils. Information was not available on how some of these alterations affect the wetland functions described in the following sections, and some impacts are hypothesized based on synthesizing other information.

4.3.1 Impacts of Changing the Physical Structure on Hydrologic Functions

No information was found on how changing the physical structure of wetlands impacts their hydrologic functions (reducing peak flows, reducing erosion, and recharging groundwater). One could hypothesize that removing erect and persistent vegetation (emergent, shrub, or forest species) may impact the reductions in water velocity that occur in wetlands. The density of vegetation is an important factor in reducing flooding or storm flows. If this vegetation is removed, the wetland will probably not be as effective at slowing these flows (in other words, there will be a change in how this wetland function is performed). As a result, downstream erosion and flooding may increase.

4.3.2 Impacts of Changing the Physical Structure on Functions that Improve Water Quality

No information was found on how changing the physical structure of wetlands affects how well wetlands remove pollutants. Removal of vegetation has impacts on both bacteria and plants, and this may affect the uptake and transformation of nutrients and toxic compounds in a wetland. The same can be hypothesized for direct alteration of soils, which may affect the chemical properties in a wetland. It is not possible, however, to predict or hypothesize how such changes might alter the wetland functions (that is, whether functions to improve water quality will increase or decrease).

4.3.3 Impacts of Changing the Physical Structure within Wetlands on Plants

By definition, removal of any vegetation causes at least a short-term change in plant biomass and possibly the composition of plant species. Vegetation can be removed by fire, tilling, mowing, or consumption of plants by animals including grazers (Newman 1991, Naiman and Rodgers 1997). Mortality from contaminants such as herbicides, logging or beaver activity, dredging or construction activities, or damage from wind, ice, or flooding can also cause loss of plants (Adamus et al. 2001).

The process by which vegetation is removed appears to influence the type, duration, and magnitude of the impact on plants. Impacts depend partly on the process through which the plants re-establish. When all or nearly all of the plants are removed through methods lethal to vegetation (such as with herbicides), recovery occurs mainly via recruitment of seeds. When removal is by non-lethal methods (such as grazing), recovery often is by vegetative regrowth. Vegetation patterns in some wetlands result in part from the differing causes of plant removal and whether those causes are lethal or not (Baldwin and Mendelssohn 1998).

The effects of grazing on wetland plants depend partly on the density of grazers, how long they are present in the grazed area, the availability of food and water in nearby alternative habitats, and the season (Clary 1995, Fitch and Adams 1998).

A study of riparian vegetation in eastern Oregon used different simulated grazing treatments to determine the effects of light and heavy grazing (Clary et al. 1996). While not clearly identified, it is evident that some of the plots were in riparian wetlands and others in non-wetland riparian habitats. The authors observed that herbaceous plants increased in growth and vigor for the ungrazed and moderately grazed plots, particularly if the grazing occurred only in the spring. Heavier, season-long grazing had detrimental effects on the vegetation.

In another study of riparian meadows in Oregon, Clary (1995) found that the biomass of the grass redtop (*Agrostis* sp.) remained stable or increased at a low-elevation site the year following simulated grazing treatments. At higher elevations, sedge species (almost all of which are found mostly in wetlands) either maintained or declined in biomass production the following year. The author concludes that grazing only annually (for several months once a year as opposed to year-round) would significantly reduce sedge production, while not decreasing redtop production.

4.3.4 Impacts of Changing the Physical Structure on Habitat for Invertebrates

The presence of invertebrate species in a wetland is influenced by the type of plants that grow there. For example, in a Washington pond, some leeches (*Helobdella*), aquatic sowbugs (*Asellus*), mayflies, and some dragonflies (especially the large-bodied *Anax*) were more commonly associated with emergent vegetation than with submerged vegetation or open water areas. Midges, freshwater shrimp (*Hyalella azteca*), and mollusks (especially *Lymnaea* sp., *Gyraulus* sp., and *Anodonta* sp.) were more common on the submerged plants (Parsons and Matthews 1995).

The removal of vegetation either mechanically or through grazing, therefore, has a significant impact on the presence and abundance of invertebrate species in a wetland. Wetland managers often manipulate the structure of vegetation by mowing, burning, plowing, or planting to encourage or discourage populations of desirable or undesirable invertebrates (Batzer and Resh 1992, Kirkman and Sharitz 1994, de Szalay et al. 1996, de Szalay and Resh 1997).

Adamus et al. (2001) conclude from their literature review that the removal of vegetation:

- Removes substrates that would otherwise provide additional vertical space in the water column for invertebrates to colonize
- Removes shade, thus increasing water temperature and causing stress for invertebrates
- Increases the circulation and perhaps the velocity of water, with accompanying increases in dissolved oxygen and possible resuspension of sediments; this may result in changes to the habitats that favor different species of invertebrates
- Reduces inputs of leaf litter that provide food to some invertebrate taxa
- Reduces structures that otherwise shelter invertebrates from predators (Jordan et al. 1994)

Complete removal of vegetation generally reduces the richness of the wetland invertebrate community, but patchy removal or moderate grazing sometimes increases richness (McLaughlin and Harris 1990, Gray et al. 1999).

4.3.5 Impacts of Changing the Physical Structure on Habitat for Amphibians and Reptiles

The information on the impacts of direct disturbances to the physical structure of a wetland on amphibians is ambiguous for Washington. In the Puget Sound Basin of Washington, surveys of 19 wetlands found no correlations that were statistically significant between amphibian richness and vegetation form (Richter and Azous 1995). Plant stem diameter is apparently more important than plant species (Richter 1997). For example, stems less than 0.1 inch (3 mm) diameter were preferred by the northwestern salamander (*Ambystoma gracile*) regardless of the plant species (Richter and Roughgarden 2002). Thus, impacts to amphibians from selective cutting or harvesting cannot be predicted.

The density of submerged plants is also important. A survey of 40 wetlands in the Puget Sound area found more native species of amphibians among wetlands containing dense emergent vegetation (Adams and Bury 1998). Dense vegetation may help protect the larvae of native aquatic amphibians from larger predators. It can be hypothesized, therefore, that removing dense emergent vegetation would probably impact the populations of amphibians.

Other studies have focused on the impacts of grazing. Based on personal observations, Maxell (2000) asserts that livestock grazing can impact amphibians through:

- Direct trampling of animals
- Trampling of vegetation that results in loss of habitat and reduces insect populations that are food sources for amphibians
- Contamination of water bodies through livestock waste
- Changes in substrate composition and bank structure
- Increased sedimentation

However, a contradictory study of the Columbia spotted frog in 127 ponds in northeastern Oregon found no significant differences between grazed and ungrazed ponds in terms of the numbers of frog egg masses and the abundance of recently metamorphosed frogs (Bull and Hayes 2000). Egg mass volume was larger at grazed sites, possibly due to a greater presence of adults or an older population (older, larger females lay bigger egg masses). Six of the eight most productive ponds (those with 20 or more egg masses) were grazed, indicating that grazing had no detrimental effect on this frog in grazed wetlands.

4.3.6 Impacts of Changing the Physical Structure on Habitat for Fish

Information in the literature did not differentiate between resident and anadromous fish. However, it does address fish in general. For example, the removal of vegetation can have a significant impact on the fish present in a wetland as a result of (Adamus et al. 2001):

- Increased water temperature that may go above the tolerance limits of certain species
- Decreased cover and thereby increased susceptibility to predation
- Changes in foods and their availability

Woody material is especially important as a source of cover for fish in off-channel wetlands such as oxbows and sloughs and in lakes (Leitman et al. 1991, Dewey and Jennings 1992, Fausch and Northcote 1992, McIntosh et al. 1994).

In lacustrine fringe wetlands, submerged plants are particularly important and their removal can change the habitat for fish. For example, declines in plants resulting from introductions of grass carp (Bain 1993) have been linked to an increase in the proportion of limnetic or open water fish species (Bettoli et al. 1991, Maceina et al. 1991, Martin et al. 1992). However, intentional thinning of plant beds can sometimes result in higher growth rates of some age classes of lake fish, presumably by giving them better access to invertebrates that are their food source (Olson et al. 1998).

4.3.7 Impacts of Changing the Physical Structure on Habitat for Birds

Many birds are sensitive to the presence and type of vegetation and its location in relationship to open water (Kauffman et al. 2001). The removal of vegetation is therefore expected to change the distribution and abundance of birds in wetlands.

Grazing has also been found to change the distribution of birds. In a study in southeastern Oregon on the effects of grazing on birds, researchers used exclosures to remove livestock from portions of riparian meadows (Dobkin et al. 1998). They found that the richness and abundance of bird species increased within the exclosures in comparison to the plots that remained available for livestock grazing. Moreover, the exclosures were dominated by wetland and riparian birds while the open plots were dominated by upland bird species.

A study of riparian habitats in Montane areas of Nevada suggests that grazing reduces the amount of suitable habitats for nesting riparian bird species (Ammon and Stacey 1999). These authors concluded that grazing reduces streamside vegetation and the diversity of vertical structure, thus making suitable nesting substrates less available. By placing artificial nests in both areas, they found that grazing facilitated nest predation, possibly because nests were more easily detected by predators in the grazed area, or because there were different predator species on each site. This study relied on artificial nests for much of the data presented on nesting success, which the authors note may be problematic. There also is a lack of sample replication in that only one area was studied for each of the two treatments.

The changes in the structure of vegetation that result from the conversion of forested wetlands to emergent and open water wetlands can alter species composition and richness of breeding birds. For example, 53% of the bird species that formerly used forested wetlands no longer occur regularly where such forests have been logged and converted to emergent wetlands (Doherty 2000 as reported in Adamus et al. 2001).

4.3.8 Impacts of Changing the Physical Structure on Habitat for Mammals

Many mammals are sensitive to the presence and type of vegetation and its location in relationship to open water. The removal of vegetation is therefore expected to change the distribution and abundance of mammals in wetlands (Adamus and Brandt 1990).

Adamus and Brandt (1990) created a synthesis of the literature on mammal habitat which serves as the basis for the following discussion.

The species richness of small mammals in wetlands has been correlated with the complexity of vegetation structure (Arner et al. 1976, Landin 1985, Nordquist and Birney 1980, Stockwell 1985, Searls 1974, Simons 1985). Removal of vegetation and associated long-term destruction of den sites in both wooded and emergent wetlands have caused changes in furbearer populations and small-mammal communities (Krapu et al. 1970, Malecki and Sullivan 1987). In contrast, restoration of riparian vegetation has led to increases in use by mink (Burgess and Bider 1980).

Grazing at levels recommended by the Natural Resources Conservation Service had no significant effect on the abundance or distribution patterns of small mammals in a cottonwood floodplain in Colorado (Samson et al. 1988).

4.3.9 Summary of Key Points

- Filling or draining a wetland can so alter the water regime that the land can no longer support wetland vegetation and maintain hydric soils. If a wetland is lost, most if not all of its functions are also lost.

- Some direct disturbances of wetlands, such as removal of vegetation, grazing, and alteration of the soil, change the wetland functions but do not result in the complete loss of functions.

- Impacts of removing vegetation on the habitat functions in wetlands have been documented for invertebrates, fish, birds, and mammals. Impacts on amphibians, however, are ambiguous. Impacts to the hydrologic and water quality functions resulting from vegetation removal can only be hypothesized since no information was found in the literature.

- Impacts of grazing on habitat functions have been documented for invertebrates and birds and are somewhat conflicting for amphibians. Impacts to fish habitat have not been studied. The one study of mammals suggests that low levels of grazing in a floodplain may have minimal impacts on the habitat of this group. No information was found on impacts of grazing on the hydrologic and water quality functions.

- No information was found on the impacts of soil alterations (through tilling and compaction) on any of the functions performed by wetlands.

4.4 Impacts of Changing the Amount of Water in Wetlands

4.4.1 Impacts of Changing Amounts of Water on Hydrologic Functions

Specific documentation was lacking on how increasing or decreasing amounts of water may affect wetland functions in reducing flooding or erosion or recharging groundwater. It can be hypothesized, however, that the storage capacity of a wetland in a depression during floods will be reduced if water levels increase. The volume that would have been available to store floodwaters is used instead to store the increased volumes coming into the wetland. This suggests that the functions related to reducing flooding would also decline because storage is a large component of flood reduction. On the other hand, wetlands in which water is deeper or covers more of the wetland may provide better recharge of groundwater because infiltration depends on the depth of water in the wetland (hydraulic head) and the area that is submerged (Hruby et al. 1999).

The converse can be hypothesized if water levels in wetlands decrease. The potential amount of water that can be stored in a wetland will increase as it becomes drier, thereby increasing the "flood reduction" functions. The function of recharging groundwater would decrease because less water would be present and it would be shallower.

4.4.2 Impacts of Changing Amounts of Water on Functions that Improve Water Quality

Some information suggests that flooding of wetlands (increased amounts of water) stimulates microbial activity, and this in turn may change how a wetland removes pollutants.

The activity of microbes potentially increases conversion of inorganic mercury to the much more toxic methyl mercury form (Kelly et al. 1997). In this case flooding would reduce the effectiveness of a wetland at improving water quality because the wetland may become a source of this more toxic compound.

Increased amounts of water may also have an impact on denitrification in wetlands. Adamus et al. (2001) reviews several studies in which the water content of soil was found to be the dominant factor controlling denitrification. In Washington, the area that is seasonally inundated was judged to be a critical factor in determining denitrification (Hruby et al. 1999). If the increase in water levels expands the area that is seasonally flooded, denitrification rates will probably increase. If, however, increases in the amount of water in a wetland expand the amount of permanent water at the expense of the areas that were seasonally flooded, the rates of denitrification can be hypothesized to decrease. Thus, wetlands in which the water regime has been changed will probably have a different rate of denitrification than they had previously. The data are insufficient, however, to predict whether denitrification rates will be higher or lower, and the change in functions depends on how the water regime is altered.

4.4.3 Impacts of Changing Amounts of Water on Plants

Much of the literature on how changing amounts of water affect plant populations in wetlands of the Pacific Northwest is in terms of changes in the dynamics of water movement (hydroperiod). This concept combines both changes in water levels and changes in how water levels fluctuate (the latter is addressed as a separate disturbance in Section 4.5.)

The composition and richness of the plant community are influenced by the saturation in the root zones of wetland plants. This is influenced by:

- The **duration** of saturation (Dicke and Toliver 1990, Merendino and Smith 1991, David 1996, Vivian-Smith 1997, Silverton et al. 1999)
- The **timing** of saturation (Merendino et al. 1990, Squires and van der Valk 1992, Scott et al. 1996, 1997, Gladwin and Roelle 1998)
- The **frequency** of saturation (van der Valk 1994, Pezeshki et al. 1996, 1998, Smith 1996, Pollock et al. 1998)

Disturbances to the dynamics of water movement and volume in a wetland can cause major changes in the distribution and richness of plant species. The response of an individual wetland to such changes, however, is difficult to predict. The existing information indicates that each plant species responds in a different way to changes in water levels. This means that overall the response of the plant community in a wetland will depend on the sum of the responses of the individual species. The following discussion summarizes some of the studies documenting how plant communities change with changes in water levels. It is beyond the scope of this document to provide detailed information on the response of individual plant species.

Responses of hundreds of plant species to specific hydrologic variables that have been studied are presented in a database at EPA's web site (Adamus and Gonyaw 2000). The database is available at http://www.epa.gov/owow/wetlands/bawwg/publicat.html

The changes in plant communities are linked to differences among plant species in their ability to resist drought and flooding. The life history and physical characteristics of plants play a role (Earnst 1990, Koncalova 1990, Voesenek et al. 1993, Kirkman and Sharitz 1993, Teutsch and Sulc 1997). The characteristics of seed dispersal and germination of plants relative to water dynamics may have the greatest effect on the relative abundance of species, according to a simulation conducted by Ellison and Bedford (1995) using six years of data from a southern Wisconsin sedge meadow. Some species, such as cattail (*Typha* spp.), are able to keep pace with rising water levels because their stem tissue elongates rapidly and to a greater degree than other species (Waters and Shay 1992, Galatowitsch et al. 1999) or they sprout adventitious roots (Voesenek et al. 1993).

Increases in inundation may change the exposure of plants to competitors and herbivores (Wilson and Keddy 1991) and cause a shift in the location of plant communities within a wetland (van der Valk et al. 1992). The opposite extreme—dehydration—kills plants partly by removing the pathway for taking up nutrients and maintaining tissues. Dehydration may also increase or decrease competition and plant exposure to herbivores (Adamus et al. 2001).

Woody plants are particularly sensitive to prolonged inundation, especially for longer than 80 days (Niswander and Mitsch 1995, Toner and Keddy 1997, Sharitz and Gresham 1997). Their seedlings consequently are most affected during years when flooding occurs at or shortly after the beginning of the growing season, or when flooding persists for more than 40% of the growing season (Toner and Keddy 1997). Annual (as opposed to perennial) species tend to increase proportionately in response to drought and some other severe disturbances (Poiani and Johnson 1989).

Species with small, light seeds seem particularly adept at colonizing mudflats exposed during drawdowns and after disturbances (Poiani and Johnson 1989, Ellison and Bedford 1995). These species tend to emerge early in the season and may be more successful by taking advantage of greater light availability (Toner and Keddy 1997).

Successive years of annual drawdowns can favor the spread of many non-native plant species within wetlands (van der Valk 1994). Dominance of a wetland by just a few species is sometimes a sign that the wetland has experienced prolonged drought or drawdown (Wilcox 1995).

Many species have only a narrow "window" in which they can germinate. For example, there may be only a few weeks when favorable water levels or a temporary lack of competitors must coincide with favorable temperatures and acceptable water quality (Rood et al. 1998).

4.4.4 Impacts of Changing Amounts of Water on Habitat for Invertebrates

Disturbances to the amount of water in a wetland can cause major changes in the distribution and richness of invertebrate species. Because each species responds in a different way to increases or decreases in water regime, the overall response of the invertebrate community in a wetland will depend on the sum of the responses of the individual species.

In general, the amounts of water in a wetland influence the distribution and richness of invertebrates by:

- Altering the amount and pattern of horizontal and vertical habitat space available for colonization (Adamus et al. 2001)
- Changing the types of algae and vascular plants that occur, the proportions of these two major food sources for invertebrates, and the seasons in which they occur (Murkin et al. 1991)
- Changing the extent of contact between plants and water, thus influencing attachment space, availability of detrital foods, shade, and shelter (Ross and Murkin 1993, De Szalay et al. 1996)
- Influencing the access of predators (Reice 1991, Martin et al. 1991, Mallory et al. 1994, Johnson et al. 1995, Wellborn et al. 1996)
- Affecting the intensity of competition (Wissinger et al. 1999)
- Causing mortality if complete desiccation or freezing occurs (Layzer et al. 1993)

4.4.4.1 Impacts of Reduced Amounts of Water on Habitat for Invertebrates

Some of the most dramatic changes to wetland invertebrate communities occur when wetlands that seldom or never dry out completely are subjected to drought or complete drawdown (Adamus et al. 2001). Less dramatic changes to invertebrate communities occur with slight alterations in the timing, duration, predictability, and depth of surface water (Eyre 1992, Giberson et al. 1992).

Drought and drawdown render the less mobile species of invertebrates more vulnerable to predation, as well as causing their direct loss due to desiccation and related factors (e.g., Stanley et al. 1994). Drought also seems to favor non-insect invertebrates, which can increase at the expense of the insect component of the invertebrate community (Hershey et al. 1999).

Coupled with the studies that show invertebrate richness increasing with longer periods of inundation, these observations indicate that removing water from a wetland may reduce the species richness of invertebrates.

4.4.4.2 Impacts of Increased Amounts of Water on Habitat for Invertebrates

An increase in the amount of water in a wetland seems to change the composition of the invertebrate community. Densities of swimming (nektonic) and bottom-dwelling (benthic) predatory invertebrates do not increase with flooding as much as the numbers of nektonic and benthic herbivores and detritivores. Predatory species can even decrease after flooding (Murkin et al. 1991), and they often increase as drought or drawdown progresses.

Although flooding generally increases the density and richness of invertebrates in wetlands, the increase may be short-lived. For example, flooding of Manitoba marshes (Murkin et al. 1991) to a level 3 feet (1 m) above normal caused a major increase in numbers of nektonic invertebrates in both vegetated and open water areas for only one year. Furthermore, densities of benthic invertebrates increased in flooded vegetation but not in open areas. The biomass of nektonic invertebrates increased only in the vegetated areas (Murkin et al. 1991).

Some researchers have observed that food webs become more complex and taxa numbers increase as wetlands become wetter, such as those that are ponded for longer periods. This has been observed in seasonal wetlands of eastern Washington (Lang 2000). Also, the use of emergence traps in 19 wetlands in King County yielded more taxa from permanently flooded than seasonally flooded wetlands (Ludwa and Richter 2000), suggesting that wetlands in which the water levels fluctuate more often will have fewer invertebrate species.

These results suggest that disturbances that cause water to remain longer in a wetland will probably increase species richness at first. The long-term effects of such increases, however, are not well understood.

4.4.5 Impacts of Changing Amounts of Water on Habitat for Amphibians and Reptiles

Most amphibians cannot tolerate prolonged dry periods. Drying of seasonal pools, especially when it occurs ahead of normal seasonal schedules, can greatly diminish the breeding success of amphibians (Rowe and Dunson 1993). This is partly because many amphibian species disperse only short distances (Berven and Grudzien 1990). Amphibian populations scattered across wetlands of varying depth and water permanence can enable species to survive long-term droughts or floods. The availability of numerous scattered wetlands can protect amphibians against effects of localized drought. Some frog and toad species living in relatively intact landscapes seem mostly unaffected, at the level of populations, by significant periods of drought (Dodd 1995).

However, amphibian populations recover slowly or not at all from droughts they might otherwise survive when habitat has become fragmented (see Section 4.11). Fragmentation results when wetlands are altered and the distance increases between remaining wetlands that are free of fish and most suitable for amphibians. Amphibian dispersal routes can be disrupted by construction of roads or other unsuitable habitats that displace terrestrial vegetation (Pounds and Crump 1994).

Both prolonged desiccation and extreme floods can increase opportunities for invasion of wetlands by exotic plant species. This can impact the suitability of a wetland as habitat for amphibians. Patterns of vegetation typically become more homogeneous, prey abundance may decline, and the habitat may become less suitable for amphibians (Ludwa 1994).

During a two-year drought in Washington, a local population of painted turtle (*Chrysemys picta belli*) suffered a 70% decline (Lindenman and Rabe 1990). This appeared to be due to both mortality and movement of turtles out of the wetland. Growth was suppressed but recovered as conditions improved. Drawing down the water level in the autumn to allow wetland management, flood control, or for other reasons can cause high mortality among juvenile overwintering turtles due to freezing if the drawdowns follow abnormally high water levels in late summer that attracted turtles (Galat et al. 1998).

These results indicate that changing the amounts of water in a wetland affects both amphibians and reptiles (specifically painted turtles). Impacts may occur from lowering the water levels (for example, through ditching, draining, or pumping) or raising the levels through increased flooding as a watershed is developed.

4.4.6 Impacts of Changing Amounts of Water on Habitat for Fish

Declines in the amounts of water alter the community structure of wetland fish. Fish experience a greater need to use overlapping resources and face an increased risk of predation when wetlands become drier (Adamus et al. 2001). Low water also increases the chance of fish freezing in winter or dying from thermal stress in summer (Adamus et al. 2001).

Sustained drawndowns can also reduce competition among fish that return to wetlands when water levels rise again by temporarily eliminating larval dragonflies and other large invertebrates that normally compete for food with the fish or prey on larval fish (Travnichek and Maceina 1994).

Impacts of increasing water levels on fish in wetlands were not documented in the literature.

4.4.7 Impacts of Changing Amounts of Water on Habitat for Birds

Disturbances to the amounts of water in a wetland can cause major changes in the distribution and species of birds. As with plants and invertebrates, the overall response of the bird community in a wetland will depend on the sum of the responses of the individual species.

4.4.7.1 Impacts of Reduced Amounts of Water on Bird Habitat

Drainage and some other disturbances in the amounts of water in wetlands have been well documented as contributing to the decline of many wetland bird species (David 1994, DeAngelis et al. 1997). In Manitoba, for example, wetland drainage has made breeding and brood-rearing areas for waterfowl less available (Rotella and Ratti 1992). As wetlands are drained or converted to other land cover types, local densities of wetlands decline and the average distances between individual wetlands increase.

Drought conditions also expose duck nests to greater predation. With drought, plants are less dense and vigorous, and islands that formerly were inaccessible gain new access points (Hallock and Hallock 1993, Jobin and Picman 1997).

Widespread drawdown of water tables reduces the number and perhaps the variety of wetlands and their vegetation communities. This in turn diminishes the richness, density, and breeding success of birds in many individual wetlands and wetland complexes (Higgins et al. 1992, Bethke and Nudds 1993, Bancroft et al. 1994, Greenwood et al. 1995, Dobkin et al. 1998).

4.4.7.2 Impacts of Increased Amounts of Water on Bird Habitat

Increasing the duration of saturation or inundation can change the use of wetlands by a variety of birds. This change can occur when shallow ephemeral ponds are dredged to make areas with longer periods of standing water (such as stock ponds). In the Columbia Basin, Creighton et al. (1997) found an increase in use by several species of diving and dabbling ducks, coots, and terns when shallow, densely emergent wetlands were dredged to create deeper pools of open water. They also documented an increase in the biomass of zooplankton, a food source for several guilds of wildlife. However, there was a decrease in use by sora (*Porzana carolina*) and Virginia rails (*Rallus limicola*) as well as red-winged blackbirds (*Agelaius phoeniceus*). The use of the excavated habitats by rails was expected to increase over time as emergent vegetation became reestablished in the excavated pools because rails prefer vegetation that is a mix of robust and thin-stemmed species. An increase in use by shorebirds was one short-term benefit. The shorebirds fed on the moist, fresh dredge spoils and exposed unvegetated soils of the newly excavated basins. Once the soils became vegetated, use by shorebirds declined.

On the other hand, while construction of reservoirs raises water levels, this affects birds by eliminating many wetlands through flooding and destabilizing water levels in the remaining wetlands (Nilsson and Dynesius 1994). Associated changes in river morphology influence the species composition of wintering waterfowl (Johnson et al. 1996).

4.4.8 Impacts of Changing Amounts of Water on Habitat for Mammals

Documentation on how disturbances to the amount of water in a wetland may affect their ability to provide habitat for mammals was not found.

4.4.9 Summary of Key Points

- Impacts of reducing water levels on the habitat functions of wetlands have been documented for invertebrates, fish, birds, and amphibians. All these groups have reduced species richness and abundance when wetlands dry up.
- Impacts of increasing water levels in wetlands on its functions as habitat have been documented for invertebrates and birds. The species richness of invertebrates may increase for a short time if a wetland becomes wetter. The impacts on the populations of birds are mixed. In some cases the richness of birds increases and in some cases it decreases.
- Impacts to the suitability of wetlands as mammal habitat resulting from either increasing or reducing water levels have not been studied.
- Reducing the amount of water changes the distribution of plants in a wetland, but the studies did not address if species richness will increase or decrease. Data suggest that woody species will tend to be replaced by more grass-like species when water levels in a wetland increase.
- Impacts to the hydrologic and water quality functions from either increasing or reducing water levels can only be hypothesized since no information on these topics was found in the literature.

4.5 Impacts of Changing the Fluctuation of Water Levels within a Wetland

A major finding of the Puget Sound Wetlands and Stormwater Management Research Program was that fluctuations in water level are key in determining biological responses. There are different types of fluctuations in water levels in a wetland and these are described in the shaded box on the following page. The researchers found a decline in the biotic diversity of wetlands associated with an increase in water level fluctuations caused by expanding impervious area within the contributing basin (Reinelt et al. 1998, Azous and Horner 2001).

Prolonged inundation (that is, less frequent water level fluctuations) resulting in a lack of oxygen in the soils has been indicated as a factor in changing the biota of wetlands. Although many hydric soils may be anaerobic, changing the length of time the soils are inundated results in prolonged anaerobic conditions and chemical changes in the soils. These changes in soil chemistry influence the survival of vegetation and soil biota that were adapted to shorter periods of inundation (Thom et al. 2001). On the other hand, key habitat elements are eliminated and biotic diversity declines in wetlands with increased periods of summer drying (Azous et al. 2001).

Mechanisms for how water level fluctuations affect aquatic systems

Richter et al. (1996) developed a method to model "indicators of hydrologic alteration" based on assessing changes in 32 hydrologic parameters they identified as being relevant to the biotic integrity of aquatic ecosystems. They divided the parameters into the following five fundamental factors that characterized how fluctuations in water levels influence biotic communities in aquatic systems:

Magnitude is a measure of the availability or suitability of habitat. It defines such habitat attributes as wetted area or habitat volume, or the position of a water table relative to wetland or riparian rooting zones.

Timing is the timing of occurrence of a particular water condition. It can determine whether certain life-cycle requirements are met. It can also influence the degree of stress or mortality associated with extreme water conditions such as floods or droughts.

Frequency refers to the frequency of occurrence of specific hydrologic conditions, such as droughts or floods. It may be tied to reproduction or mortality events of various species, thereby influencing population dynamics.

Duration is the length of time over which a specific hydrologic condition exists. It may determine the success of a particular species' life cycle or the accumulation of stressful effects.

Rate of change in hydrologic conditions may be linked to stranding of individuals (in isolated pools or along a wetted edge). It may also be related to the ability of sensitive species to maintain root contact within the phreatic zone (the portion of the soil that is influenced by proximity to the groundwater table).

4.5.1 Impacts of Changing Fluctuations in Water Levels on Hydrologic Functions

The literature did not provide explicit information on possible impacts of changes in water level fluctuations on the hydrologic functions of wetlands. It is not possible at this stage to hypothesize either positive or negative impacts on hydrologic functions. The major questions that need to be addressed include:

- Will changes in the frequency or amplitude of water level fluctuations change the flood storage capacity of a wetland?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland reduces water velocity?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland recharges groundwater?

4.5.2 Impacts of Changing Fluctuations in Water Levels on Functions that Improve Water Quality

How changing fluctuations in water levels impact the ability of wetlands to improve water quality was not detailed in the literature. It is not possible to hypothesize either positive or negative impacts on water quality functions. The major questions that need to be addressed include:

- Will changes in the frequency or amplitude of water level fluctuations change how a wetland traps sediment?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland removes nitrogen?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland captures or transforms toxic compounds?

4.5.3 Impacts of Changing Fluctuations in Water Levels on Plants

In general, the amplitude and rate of water level fluctuation have been found to influence the species composition, biomass, and germination of plants (Hudon 1997, Shay et al. 1999). Furthermore, the timing of inundation and duration throughout the seasons also influences plant species richness and survival (Ewing 1996, Reinelt et al. 1998, Owen 1999, Azous et al. 2001).

Researchers consistently found a decline in plant species richness in urbanized watersheds where water level fluctuations had increased (Azous and Cooke 2001).

Among 26 wetlands in the Seattle area, the degree of seasonal water level fluctuation was negatively associated with richness found in emergent and shrub wetlands, but it had no statistically significant effect on species richness in the forested wetlands (Cooke and Azous 2001). These authors found that fluctuation during the early spring seemed to have an especially detrimental effect on plant richness in the emergent and shrub wetlands.

Reinelt et al. (1998) found that the development of plant communities in lowland wetlands of Puget Sound was related to water level fluctuations and depth of inundation during the early growing season. They noted that shifts in the “hydrologic profile” of the wetland caused a subsequent shift in the species composition of the wetland’s plants. The emergent and scrub-shrub communities of the wetland tended to have lower plant richness when average annual water level fluctuations increased to over 8 inches (20 cm).

Azous and Horner (2001) determined that the duration of flooding, as well as depth, also strongly influenced plant diversity. They noted greatest plant diversity when:

- Flooding events were less than 0.5 feet (0.2 m) above predevelopment levels
- Floods were limited to an annual average of three or fewer events per month
- The cumulative duration of flooding was less than six days per month above predevelopment averages

On the other hand, a lack of water level fluctuation can be just as damaging as excessive fluctuation to some wetland species (Rood and Mahoney 1990). This is because many species need a period of desiccation in order to germinate.

Evidence from some studies suggests that the relative tolerance to water level fluctuations is greatest among several non-native or invasive species (Figiel et al. 1991, Haworth-Brockman and Murkin 1993, King and Grace 2000). Increases in water level fluctuations and duration of inundation favor generalist plants (plants that are found under a wide range of environmental conditions) in the Pacific Northwest (Azous et al. 2001).

These results indicate that changes to water level fluctuations in wetlands will change the plant species present in the wetland. Furthermore, increases in water level fluctuations will probably facilitate the invasion of non-native or “aggressive” native species.

4.5.4 Impacts of Changing Fluctuations in Water Levels on Habitat for Invertebrates

In the Northwest, researchers have observed a decline in the diversity of invertebrates with an increase in impervious area in the basin, which to a large degree results in changes in the fluctuations of water levels (Ludwa 1994, Hicks 1996, Ludwa and Richter 2001a, Thom et al. 2001). Information from other parts of the United States seems to confirm this.

The densities of some invertebrate species can be decimated by rapid water level fluctuations, especially when the fluctuations are more frequent and severe than those historically encountered in the wetland. For example, Missouri floodplain pools that experience fluctuations in water level at extreme frequencies and amplitudes tend to have lower invertebrate density (Magee et al. 1993). Repeated exposure to desiccation in a short period of time can lead to a marked reduction in the density of invertebrates. In an Arizona stream that experienced 12 flash floods between August and December of a single year, densities of all invertebrates were reduced by 75 to 100% (Boulton et al. 1992). In particular, the numbers of water spiders, midges, and some caddisflies, mayflies, and snails declined.

In contrast, some groups of invertebrates appear quite resilient to periodic fluctuations. In the Arizona study referenced above (Boulton et al. 1992), the oligochaete (worm) populations appeared to be unaffected. In a British Columbia river, populations of the mayflies *Rhithrogena* and *Baetis*, as well as the caddisfly *Hydropsyche* (species found in wetlands as well) survived flows that increased rapidly during flooding from 500 to 6,500 cubic meters per second (Rempel et al. 1999). In an Oklahoma intermittent stream where spring and fall floods reduced the density of invertebrates by 90%, the mayflies *Caenis* sp., *Leptophlebia* sp., and *Baetis* sp. were especially resilient and midges were less so (Miller and Golladay 1996).

A number of studies have found that reducing fluctuations in streams by maintaining minimum water levels (such as in reservoirs) can increase invertebrate densities in the part of an adjacent wetland that is not permanently inundated (Weisberg et al. 1990, Troelstrup and Hergenrader 1990).

4.5.5 Impacts of Changing Fluctuations in Water Levels on Habitat for Amphibians and Reptiles

In Puget Sound wetlands, amphibian species richness was negatively correlated with the percent of impervious cover in a contributing basin. The primary cause is increased water level fluctuation (Richter and Azous 2001a). The richness of amphibians declined to less than three species when water level fluctuations increased to over 8 inches (20 cm) (Richter and Azous 2001a, Thom et al. 2001). Chin (1996) concluded that the reduced richness of amphibians was correlated with a reduction in the diversity of wetland plants that resulted from increases in water level fluctuations.

Increases in fluctuation of water levels also affect amphibians by (1) stranding egg masses when water levels drop, and (2) reducing the thin-stemmed emergent plant species on which amphibians lay their eggs. Previous unpublished work by Richter and Roughgarden (2002 in press) in western Washington found that amphibians preferred thin-stemmed vegetation on which to lay their egg masses. Greater water level fluctuation directly affects amphibian egg survival and causes changes in plant species, reducing the thin-stemmed emergent species used by amphibians for egg laying (Chin 1996, Richter and Roughgarden 2002 in press).

No correlations were found between the richness of amphibian species and a variety of other factors including wetland size, distance to breeding habitats, presence of predators, and number of vegetation classes (Richter and Azous 2001a). The most significant factor affecting species richness was mean water level fluctuation, with 8 inches (20 cm) mean annual fluctuation being a threshold for lentic breeding species (those that breed in stagnant or slow-moving waters such as ponds and wetlands). Lentic breeding amphibians appear to be affected by increases in the duration and frequency of flooding and increased discharge rates resulting from the greater frequency and magnitude of storm peaks in urban watersheds (Richter and Azous 2001a).

Amphibian populations in western Washington generally experience impacts in contributing basins with more than 10% impervious surface area (Booth and Reinelt 1993). A more recent study documented that watersheds with less than 15% total impervious area had three or more amphibian species, whereas most watersheds with more than 25% impervious area had less than three species (Chin 1996). Chin (1996) concludes that changes in water level fluctuations and maximum water levels during spring breeding and embryo development are the primary adverse effects of increased impervious surface.

4.5.6 Impacts of Changing Fluctuations in Water Levels on Habitat for Fish

Researchers compared the use of two watersheds in King County by two species of fish, coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*Salmo clarki*). They identified a “marked degradation” in relative fish use at 8 to 10% total impervious area within the watershed (Lucchetti and Fuerstenberg 1993 as cited in Booth and Reinelt 1993). It can be assumed that much of this impact is a result of changes in water level fluctuations because this is one of the major impacts of impervious surfaces in a contributing basin.

4.5.7 Impacts of Changing Fluctuations in Water Levels on Habitat for Birds

General observations have indicated a decline in bird richness for wetlands located in a contributing basin that is developed or developing. Richness was not reduced in contributing basins that remained rural or relatively undeveloped over the course of the Puget Sound Wetlands and Stormwater Management Research Program (Richter and Azous 2001b, Thom et al. 2001). However, these observations have not specifically been correlated with changes in the fluctuation of water levels.

4.5.8 Impacts of Changing Fluctuations in Water Levels on Habitat for Mammals

No explicit information on how changing fluctuations in water levels will impact mammal populations in wetlands was presented in the literature. It is not possible to hypothesize either positive or negative impacts on mammal populations.

4.5.9 Summary of Key Points

- No information was found on the impacts to the hydrologic and water quality functions of wetlands resulting from altered fluctuations in water levels.
- Changes in how water levels fluctuate in wetlands have documented impacts on invertebrate and amphibian habitat. Both groups of wildlife exhibit reduced species richness and abundance when wetlands are subject to increased fluctuations in water levels. Impacts to the suitability of wetlands as habitat for mammals, fish, and birds have not been documented.
- Increasing fluctuations in water levels also reduce plant richness in wetlands.

4.6 Impacts of Changing the Amounts of Sediment

4.6.1 Impacts of Changing Amounts of Sediment on Hydrologic Functions

Despite a lack of explicit information on impacts that sedimentation may have on hydrologic functions, it is possible to hypothesize that increases in sediment load to a wetland will reduce the amount of water it can store. For every cubic yard of sediment deposited in a wetland, the storage capacity of water is reduced by a similar amount. This means that wetlands along stream corridors with high inputs of sediment may lose much of their ability to store surface waters during floods. Some wetlands with a lot of erosion in the contributing basin, but not along stream corridors, may also have high inputs.

This is especially true for depressional wetlands. By definition, depressional wetlands function to reduce flooding by storing water behind a restricted outlet and then releasing it slowly. There is less chance in depressional wetlands than in riverine wetlands that severe floods will erode the new sediments and restore the wetland’s storage capacity.

4.6.2 Impacts of Changing the Amount of Sediment on Functions that Improve Water Quality

Whether changing the sediment load to a wetland has positive or negative impacts on the water quality functions is not documented in the literature.

4.6.3 Impacts of Changing the Amount of Sediment on Plants

Accelerated sediment deposition or erosion can tax the ability of plant communities to adapt (Kantrud et al. 1989, Jurik et al. 1994, Wang et al. 1994). Sediments have been found to impact plant communities in wetlands in several general ways:

- **Burying seeds, leaves, or plants.** Sedimentation can bury established vegetation and seed banks (Adamus et al. 2001). The burial of leaves prevents photosynthesis and restricts gas exchange through foliage (Ewing 1996). Buried plants expend energy elongating their shoots in an attempt to outpace sedimentation, seeking oxygen and light, and consequently may be less robust.

- **Changing the depth of habitats.** Over the long term, sedimentation can shrink shallow wetlands or reduce the depth of ponds that previously were too deep to support many wetland plants. Such long-term changes in water depth or relative elevation also result in shifts in species composition, as has been documented in the Mississippi River floodplain (Adamus et al. 2001).

- **Inhibiting germination.** Seeds of the most sensitive species often fail to germinate when buried (Dittmar and Neely 1999). The addition of sediment has been found to reduce germination rates of wetland herb species by 34% (Neely and Wiler 1993), 80% (Jurik et al. 1994), and 90% (Wang et al. 1994) depending on the species involved. In general, the species with larger seeds appear to be better able to survive burial (Dittmar and Neely 1999, Jurik et al. 1994, Wang et al. 1994).

  Less than 0.5 inch (1 cm) of sediment can inhibit germination of cattails (*Typha* sp.), barnyard grass (*Echinocola crusgalli*), rice cutgrass (*Leersia oryzoides*), and sedges (*Carex* sp.) (Jurik et al. 1994). Sedimentation inhibits the germination of cattail (*Typha latifolia*) seeds more than seeds of bur-reed (*Sparganium eurycarpum*) (Neely and Wiler 1993). Germination of cattail (*Typha* x *glauca*) seeds decreased by 60 to 90% when sediment loads of less than 0.5 inch (0.2 to 0.4 cm) were applied to the surface of the soil (Wang et al. 1994).

  In contrast, burial by 1 inch (2 cm) of sediment does not interfere with germination of several non-native plant species (Blackshaw 1992, Reddy and Singh 1992).

- **Reducing survival of seedlings.** Excessive sedimentation can reduce the survival of seedlings (Jurik et al. 1994). For example, the density of cattail seedlings and their biomass decreased as sediment loads increased from 0.08 to 0.5 inch (0.2 to 1.0 cm). One study found a fourfold greater density of annuals (vs. perennials) in some heavily sedimented sites (Neely and Wiler 1993). Older and larger seedlings were more tolerant of burial (Wang et al. 1994).

- **Favoring species more tolerant of sediment.** Sedimentation impacts individual wetland species in different ways. The composition of the plant community will therefore change as the most sensitive species are suppressed by the sediments while the more tolerant ones thrive. Effects of sedimentation on particular wetland plant species are not well documented (van der Valk and Jolly 1992) but findings relevant to wetland species found in Washington are discussed here.

  Many mature plants, and especially woody species, apparently are not harmed by a small amount of sedimentation (Wang et al. 1994). Adult plants of wild celery (*Vallisneria americana*) tolerated burial to depths of up to 4 inches (10 cm) but none survived burial under sediment depths of 10 inches (25 cm) (Rybicki and Carter 1986). Among woody plants, saplings of red alder (*Alnus rubra*) tolerated burial less well than those of Oregon ash (*Fraxinus latifolia*) (Ewing 1996).

  Growth of the invasive reed *Phragmites australis,* however, typically keeps pace with moderate rates of sedimentation (Pyke and Havens 1999). However, seeds, seedlings, and plants that have evolved in wetland types in which sedimentation is rare (such as bogs) are highly sensitive to burial. The size of particles that are being deposited, not just their amount, may also influence plant survival (Dittmar and Neely 1999).

4.6.4 Impacts of Changing Amounts of Sediment on Habitat for Invertebrates

In general, increased amounts of sediment can reduce the richness and density of invertebrates and alter their species composition. Excessive sedimentation affects invertebrates through several mechanisms (reviewed in Adamus et al. 2001):

- Burial of detritus and algae, which are important food sources

- Increase in the time required for invertebrates to move through deposited sediment and collect scarce food items

- Reduced flow of water through soil particles, which is necessary to supplying invertebrates with adequate dissolved oxygen

- Mortality of plants that otherwise provide attachment structures and shelter to invertebrates

Some studies have linked changes in invertebrate communities to the development of watersheds (e.g., Hogg and Norris 1991, Ludwa 1994, Carlisle et al. 1998, Ludwa and Richter 2000a). Development often is accompanied by increased export of sediment to water bodies.

Many invertebrate communities in wetlands are adapted to occasional deposition of small amounts of sediment, whereas constant or severe deposition causes major changes. The following bullets summarize some of the studies that have documented impacts of sediment on individual invertebrate species, as well as groups of species, many of which are found in Washington.

- Once deposited, sediments can further damage wetland invertebrate communities if they are resuspended by wind mixing or fish, making the water turbid. For example, bottom-feeding carp (*Cyprinus carpio*) noticeably increase turbidity, both directly (as they move along the bottom) and by consuming aquatic plants that otherwise would stabilize and trap sediments (Lougheed et al. 1998). The biomass of planktonic invertebrates declined in Utah ponds after introduction of carp (Huener and Kadlec 1992).

- In some instances, invertebrate density and perhaps richness can increase over the long term if sedimentation replaces coarser substrates with finer substrates that better support establishment of rooted plants. In temporarily flooded prairie pothole wetlands, only caddisflies seemed relatively unaffected by surrounding land use that generated sediments. Ostracods (seed shrimp), cladocerans (water fleas), and some snails (planorbiids, lymnaeids, physids) were diminished, presumably in part because of sedimentation (Euliss and Mushet 1999).

- Burrowing, tube-forming worms and midges commonly predominate where sediments accumulate (Magee et al. 1993). Filter-feeding species and those that graze on the bottom are most sensitive (Lougheed and Chow-Fraser 1998). However, invertebrate size and behavior also influence their tolerance to sediments (McClelland and Brusven 1980). On the other hand, substrates newly created by sedimentation may attract tolerant individuals and species that are poor competitors on older, more crowded substrates (Soster and McCall 1990).

- Severe and rapid sedimentation is inevitably lethal to nearly all aquatic invertebrates. In North Dakota, wetlands surrounded by cropland were virtually devoid of the resting eggs of zooplankton, whereas such eggs were abundant in wetlands surrounded by mostly natural grassland, which presumably minimized erosion and sedimentation (Euliss and Mushet 1999).

- Unionid mussels (mussels in the family Unionidae) are sensitive to increased sedimentation (Goudreau et al. 1993, Box and Mossa 1999). Numbers of the swamp fingernail clam (*Musculium partumeium*) and amphipods were reduced in willow wetlands in northeastern Missouri where 2 to 4 inches (5 to 10 cm) of sediment had been recently deposited (Magee et al. 1993).

- Sediments may clog the filter feeding mechanisms of some species and limit light penetration. This would adversely impact phytoplankton and other primary producers, with a subsequent adverse impact on food chains (Euliss and Mushet 1999).
- Sedimentation also potentially buries invertebrate eggs deposited in the substrates of wetlands (Euliss and Mushet 1999).

4.6.5 Impacts of Changing Amounts of Sediment on Habitat for Amphibians and Reptiles

Few studies of the impacts of increases in the deposition of sediment on amphibians and reptiles have been conducted in wetlands. On one hand, some species require soft sediments as hibernation sites. For example, painted turtles (*Chrysemys picta*) used sediments 1.6 to 3 feet (0.50 to 0.95 m) thick in an Ontario pond (Taylor and Nol 1989). On the other hand, excessive sediments, when stirred, impair light penetration of the water column and thus can inhibit growth of algae and especially submersed aquatic plants, which provide cover and attachment sites for amphibian eggs.

4.6.6 Impacts of Changing Amounts of Sediment on Habitat for Fish

No recent studies on the impacts of sediment on habitat for fish in North American wetlands or lakes were found. Most of the studies on the impacts of sediment on fish populations have been done in streams, especially as it concerns the growth and reproduction of salmonids in the Pacific Northwest. This information was reviewed and synthesized in Knutson and Naef (1997). The conclusion reached by Knutson and Naef quoted below can also apply to wetlands because streams are often considered a part of wetlands:

> *Sedimentation in fish-bearing waters affects habitat quality and fish survival in a number of ways. Stream bottoms covered with fine sediments are no longer suitable for spawning. Sediments cover and suffocate fish eggs and fry. High sediment deposits also block fish passage to upper spawning reaches. Suspended sediments clog the gills of fish, decrease dissolved oxygen levels, inhibit fish feeding and growth, and suppress macroinvertebrate food sources.*

4.6.7 Impacts of Changing Amounts of Sediment on Habitat for Birds

Little information was found on how sedimentation impacts the habitat that a wetland provides for bird communities. One can hypothesize, however, that sedimentation will impact birds by altering habitat structure, killing submersed vegetation, or altering the abundance or availability of prey items. In one case, however, densities of breeding dabbling ducks were positively correlated with wetland turbidity in ponds in the interior of British Columbia (Savard et al. 1994).

4.6.8 Impacts of Changing Amounts of Sediment on Habitat for Mammals

How sedimentation impacts the habitat that a wetland provides for mammals was not documented in the literature. As with birds, however, one can hypothesize that sedimentation can impact mammals by altering habitat structure or changing the abundance or availability of prey items.

4.6.9 Summary of Key Points

- Impacts to the hydrologic functions from sedimentation can be hypothesized because an increase in sediments reduces the storage capacity of a wetland.
- No information was found on possible impacts of sedimentation on the functions of wetlands that improve water quality.
- Increasing sedimentation will also decrease plant richness and tends to favor the more invasive types that tolerate disturbance.
- Impacts of increased amounts of sediment on the habitat functions of wetlands have been documented for invertebrates, amphibians, and fish. All of these groups generally have reduced species richness and abundance when wetlands are subject to increased sedimentation. In some cases, however, where the sediments coming into a wetland are finer than existing sediments, the number of invertebrate species may increase. Impacts from sedimentation on the suitability of wetlands as habitat for mammals and birds have not been documented.

4.7 Impacts of Increasing the Amount of Nutrients

The major nutrients for plant growth, phosphates, nitrates, and ammonium, can be transported into aquatic systems in dissolved forms or adsorbed onto sediment. The element phosphorus is usually the limiting nutrient for plant growth in freshwater aquatic systems (Newton 1989, Mitsch and Gosselink 2000). Because it is the limiting factor, phosphorus in the presence of the other critical element, nitrogen, allows expansive growth of phytoplankton, algae, and larger plants in aquatic systems when it is available in higher quantities.

Excessive algal growth is unsustainable, and when the algae blooms die, their decomposition causes the available dissolved oxygen to be consumed. This cycle of excessive plant growth, plant death, and decomposition that uses up oxygen is called *eutrophication*.

Excess phosphorous and nitrogen, therefore, often leads to eutrophication with subsequent mortality of the aquatic organisms that require oxygen (Newton 1989, Mitsch and Gosselink 2000). Wetlands with areas of water on the surface can therefore become eutrophic if they receive excessive amounts of phosphorus and/or nitrogen.

4.7.1 Impacts of Increasing Amounts of Nutrients on Hydrologic Functions

It is possible that the stimulation of plant growth by excess nutrients could increase the density of plants in the wetland. A thicker stand of vegetation can be expected to provide more resistance to flood flows than a thinner one (Hruby et al. 1999). Therefore, excess nutrients might indirectly improve the reduction in velocity that a wetland provides during floods. The literature did not provide any other information on how nutrient impacts might affect the hydrologic function of wetlands.

4.7.2 Impacts of Increasing Amounts of Nutrients on Functions that Improve Water Quality

Some research indicates that excessive nutrients from agricultural operations may reduce the normal ability of wetland microbes to detoxify particular pesticides (Kazumi and Capone 1995, Chung et al. 1996, Entry and Emmingham 1996). Adding nitrogen to riparian wetlands may potentially compromise the long-term ability of the system to remove nitrogen via denitrification (Ettema et al. 1998). Other information on this topic was not documented in the literature.

However, several avenues of research could be combined to make some hypotheses about impacts. The addition of nutrients to acidic bogs results in changes in plant communities. The plant community that maintains the high acidity in the bog may change to one that maintains a more neutral pH. These changes might then alter several aspects of chemistry in the wetland that affect its ability to improve water quality. The rate of nitrification will probably increase because, as noted by Mitch and Gosselink (2000), low pH inhibits denitrifying bacteria. The change in pH will also probably change the ability of the wetland to bind different toxic metals and other compounds. (See the discussion in Chapter 2 on how pH is linked to the ability of a wetland to bind different pollutants.)

4.7.3 Impacts of Increasing Amounts of Nutrients on Plants

Excessive nutrients can affect wetland plants in a variety of ways including:

- Shifting the species composition away from species that take up nutrients slowly, to those that are able to exploit nutrient pulses more rapidly or which have high nutrient requirements (Hough et al. 1989, Arts et al. 1990, Gopal and Chamanlal 1991, Wetzel and van der Valk 1998)

- Triggering algal blooms that can shade out many submersed herbaceous plants (Crowder and Painter 1991, Stevenson et al. 1993, Srivastava et al. 1995, Short and Burdick 1995)
- Causing dead plant material to accumulate faster than it can decompose completely, thus altering understory and soil structure (Neill 1990b, Craft and Richardson 1993)

Such changes usually result in long-term changes in the distribution and richness of plants within the wetland. Over the long term, nutrient additions to most wetlands tend to reduce species richness and increase the dominance of a few species. Often, non-native species are most capable of invading rapidly changing environments. Consequently they frequently come to dominate some nutrient-enriched wetlands (Adamus et al. 2001).

Increases in plant litter can smother other plants when the fast growing species die, thus helping maintain the dominance of species that exploit nutrients the most (Adamus et al. 2001). For example, the addition of nitrogen and phosphorus fertilizers to a marsh dominated by cattail (*Typha glauca)* and the grass *Scolochloa festucacea* during two growing seasons resulted in increased biomass of both species. However, the biomass of *S. festucacea* declined in the second year due to accumulated litter of *T. glauca* (Neill 1990b).

The plants in bogs and other nutrient-poor wetlands are logically the most sensitive to nutrient additions (Moore et al. 1989). The increased availability of nutrients allows grasses and common opportunistic plants to outcompete rare plants (such as sundews, orchids, and pitcher plants) that are adapted to nutrient-poor conditions. For example, in Appalachian peat bogs, the spatial dominance of bristly dewberry (*Rubus hispidus*) was positively related to nutrient levels, but dominance of the Ericaceae shrubs was negatively related (Stewart and Nilsen 1993).

Many aquatic plant species respond to nutrient additions with increased growth, biomass, and productivity. Growth responses to enrichment have been documented for about 80 wetland-associated species in North America. Of these, most have tolerated enrichment or responded to enrichment with increased biomass or growth (Adamus and Gonyaw 2000).

> Information on the response of many individual plant species to nutrients can be found in the National Database of Wetland Plant Tolerances at:
> http://www.epa.gov/owow/wetlands/bawwg/publicat.html#database1

4.7.4 Impacts of Increasing Amounts of Nutrients on Habitat for Invertebrates

Excessive nutrients can cause long-term and short-term shifts in invertebrate communities. The information available suggests that excess nutrients can result in both decreases and increases in species richness as well as changes in the groups of invertebrates found. The direction of the change depends on how the nutrients impact the vegetation and soils that are the main habitat for invertebrates. Findings from the literature include:

- **Increased richness of invertebrates.** Up to some point, nutrient inputs to wetlands can lead to increased invertebrate richness, as more food sources become available to predatory invertebrates (Rader and Richardson 1992, Campeau et al. 1994, Cieminski and Flake 1995, Gernes and Helgen 1999).

- **Reduced richness of invertebrates.** Invertebrate richness in a series of highly enriched wastewater wetlands was found to be lower than in a less enriched reference wetland (Nelson et al. 2000).

- **Changes in the types of invertebrates.** In some cases excess nutrients result in the increased dominance of certain kinds of algae. Invertebrates that specialize in feeding on these algae, or that characteristically find shelter and attachment sites in the aquatic plants, then have an advantage and can become dominant (Murkin et al. 1991, Campeau et al. 1994). Exposure to organic enrichment and eutrophication frequently causes an increase in grazers (such as Tanypodinae midges), as well as other herbivores, species that feed on detritus, predators, and "miners" that burrow into plants. These are groups that typically increase with increasing growth of algae growing on the bottom and emergent aquatic plants (Campeau et al. 1994). A study of four lacustrine/bay wetlands bordering Lake Michigan also found that midge communities shifted across nutrient gradients (Murkin et al. 1994, Campeau et al. 1994).

- **Increased density of invertebrates.** Total invertebrate density increases with increased nutrients, as algal production becomes less of a limiting factor in the invertebrate community (Murkin et al. 1992, Campeau et al. 1994).

- **Changes in the bioaccumulation of metals by invertebrates.** Nutrients appear to influence the tendency of aquatic invertebrates to accumulate heavy metals and the type of metals that are accumulated. For instance, zinc, iron, and manganese concentrations were higher in midges from nutrient-rich wetlands, whereas high copper concentrations were found in midges from nutrient-poor wetlands (Bendell-Young et al. 1994). This may be due at least partly to the bioavailability of various metals being influenced by oxygen conditions in the sediment, which in turn are partly the result of decomposition of algal blooms triggered by high concentrations of nutrients (Adamus et al. 2001).

4.7.5 Impacts of Increasing Amounts of Nutrients on Habitat for Amphibians and Reptiles

The review of the literature indicates that amphibians can be impacted by the input of nutrients. No studies were found on impacts on reptiles.

Amphibians in the Northwest can be directly impacted by the input of nitrates. Five amphibian species in Oregon showed both sublethal responses and mortality following laboratory applications of nitrate. These studies indicated that the EPA nitrate criteria for drinking water of 10 milligrams per liter (mg/l) and/or for protection of warmwater fish are inadequate to protect these amphibians (Marco et al. 1999). In Texas, playa wetlands receiving nutrient-laden effluent from feedlots were devoid of amphibians found in natural playas (Chavez et al. 1999). Experiments indicated that effluent had to be diluted to less than 3% strength in order to minimize adverse effects on the leopard frog (*Rana pipiens*).

Indirect impacts of excessive nutrients can also be important to amphibians. Shifts in seasonal timing and amount of nutrients that enter a wetland can, over a period of years, increase the relative dominance of algae and/or emergent plants at the expense of submersed plants. This in turn can reduce the availability of submersed plants as attachment substrates for amphibian eggs and as cover for larvae (Beebee 1996).

Excess nutrients can also diminish dissolved oxygen levels (Tattersall and Boutilier 1999), alter the abundance of aquatic predators, and shift the algal and invertebrate foods available to amphibians (Horne and Dunson 1995). As a result, species composition and sometimes species richness of amphibian communities can decline as eutrophication becomes severe. However, well designed studies of such effects are few.

4.7.6 Impacts of Increasing Amounts of Nutrients on Habitat for Fish

Direct impacts of excess nutrients on fish in wetlands were not documented in the literature. However, the secondary impacts of eutrophication such as oxygen depletion do affect fish. Much of the literature deals with impacts of low oxygen in streams (for a review see Knutson and Naef 1997), and it can be assumed that the impacts of low oxygen in wetlands will be similar.

As mentioned previously, the increased plant production that results from added nutrients often results in low oxygen levels when the plant material dies and starts to decompose. Many fish species suffer from reduced levels of dissolved oxygen, and feeding habits also may shift. To some degree, fish families can be grouped according to their susceptibility to oxygen deficiencies. Salmonids and coregonids (whitefish) require high levels of dissolved oxygen, whereas cyprinids (a large family that includes carp and goldfish) often tolerate low dissolved oxygen levels (Harper 1992). Thus the species composition and richness may change depending on the initial state of the wetland and the duration and magnitude of the eutrophication.

4.7.7 Impacts of Increasing Amounts of Nutrients on Habitat for Birds

Eutrophication can indirectly impact the composition of the wetland bird community by altering the vegetation structure and availability of prey. In general, moderately elevated nutrient levels also spur the growth of submersed plants that provide food for ducks, as well as supporting more aquatic insects that are especially important as food for ducklings and aerial foragers like swallows. However, excessive nutrients cause algal blooms that can kill fish eaten by birds, reduce the growth of plants growing on the bottom by blocking light, and reduce visibility of other food items under the water surface.

Studies that have documented changes in the bird community related to excess nutrients are summarized below:

- Excessive nitrates have been implicated in deaths of some frogs (see Section 4.7.5). Frogs are significant prey item for many wetland birds (Adamus et al. 2001).
- Northern shoveler (*Anas clypeata*) and eared grebe (*Podiceps nigricollis*) were positively associated with phosphorus in a survey of wetlands in interior British Columbia (Savard et al. 1994).
- Water-bird abundance and biomass were positively correlated in 46 Florida lakes with levels of phosphorus, nitrogen, and chlorophyll. There also was a positive correlation of water-bird richness with phosphorus, after accounting for nutrients contributed to the lakes by the birds themselves (Hoyer and Canfield 1994).
- Total density of dabbling ducks was correlated positively with total dissolved nitrogen (Savard et al. 1994).
- The parasitic nematode *Eustrongylides ignotus,* which has only been found in disturbed and enriched wetlands (Spaulding and Forester 1993), negatively affects the health of adult wading birds and the survival of their nestlings (Spaulding et al. 1993).

4.7.8 Impacts of Increasing Amounts of Nutrients on Habitat for Mammals

Impacts from increases in nutrients on the habitat of mammals in wetlands are not described in the literature. It can be hypothesized, however, that if eutrophication results in anoxic conditions that are lethal to the prey of mammals (e.g., fish and some amphibians), the community composition may shift from predator species (such as otter or mink) to vegetarian or invertebrate-eating species and opportunists (such as muskrat).

4.7.9 Summary of Key Points

- Some impacts to the hydrologic functions from increased nutrients can be hypothesized because the increased growth of plants resulting from increased nutrients may provide better resistance to the movement of flood waters.
- Some impacts to the functions of improving water quality have been reported. These include a potential reduction in the ability of wetlands to detoxify pesticides and to remove nitrogen as a pollutant. Impacts from increased nutrients can also be hypothesized for bogs. The ability of bogs to bind toxic metals may be reduced but their ability to remove nitrogen may be increased.
- Increasing nutrients will stimulate plant growth and may change the composition of the species present.
- Impacts of increased amounts of nutrients on the habitat wetlands provide have been documented for invertebrates, amphibians, and birds. Excess nutrients can result in both an improvement in the habitat through the production of food and a reduction in habitat through eutrophication. The actual impacts depend on local conditions in the wetland. Impacts to the habitat for fish and mammals can be inferred because eutrophication causes reductions in the levels of oxygen in the water with resultant impacts to both water quality and the food sources for these two groups.

4.8 Impacts of Increasing the Amount of Toxic Contaminants

4.8.1 Impacts of Increasing Amounts of Toxic Contaminants on Hydrologic Functions

No explicit information was found in the literature on the possible impacts of toxicity from contaminants on the hydrologic functions provided by wetlands (storing flood waters, reducing erosion, and recharging groundwater).

4.8.2 Impacts of Increasing Amounts of Toxic Contaminants on Functions that Improve Water Quality

Information on how toxic compounds affect the function of wetlands to remove pollutants is sparse. It can be hypothesized, however, that an input of low levels of toxic compounds may stimulate the ability of a wetland to detoxify pollutants. Some microbial species biodegrade particular contaminants and their abundance is increased in the presence of low levels of the contaminants. These species can flourish in some wetlands that are only mildly or moderately contaminated.

Contaminants that can be processed by microbes when at low to moderate concentrations include copper (Farago and Mehra 1993), mercury (Marvin-Dipasquale and Oremland 1998), selenium (Steinberg and Oremland 1990, Azaizah et al. 1997), cadmium (Sharma et al. 2000), manganese (Sikora et al. 2000), and petroleum (Nyman 1999, Megharaj et al. 2000).

4.8.3 Impacts of Increasing Amounts of Toxic Contaminants on Plants

Most plant species are relatively tolerant to toxic contaminants. Impacts usually result from the effects of contaminants on plant metabolic pathways, enzymatic reactions, and growth (Fitter and Hay 1987). Symptoms of toxicity can include reduced growth; small, discolored, or dying leaves; early leaf fall; and stunted or suppressed growth of roots (Pahlsson 1989, Rhoads et al. 1989, Vasquez et al. 1989).

Shifts in the composition of the plant community in response to contaminants have not been widely documented. Relevant studies include:

- Arsenic, cadmium, copper, lead, and zinc inhibited growth in hybrid poplar (*Populus*) and several other tree species (Lejeune et al. 1996).
- Iron and manganese, although not usually toxic to wetland plants, do affect species in some wetland types. For example, laboratory experiments revealed differences among 44 fen species with regard to the influence of iron on growth (Snowden and Wheeler 1993).
- Oil spills can have long-lasting effects on wetland plant communities (Obot et al. 1992). In a greenhouse experiment, oil and a detergent used to clean up oil spills were applied to broadleaf arrowhead (*Sagittaria lancifolia)*, salt marsh sedge (*Scirpus olneyi),* and common cattail (*Typha latifolia)*. The leaves on all of the study plants died following oiling, but new leaves soon developed on those plants subjected to oil and subsequent cleaning with the detergent. *S. olneyi* was the least sensitive of the three species, whereas *T. latifolia* appeared to be the most sensitive (Pezeshki et al. 1998).
- The herbicides Rodeọ® and Garlan 3A®, applied to control purple loosestrife (*Lythrum salicaria*), also reduced the growth rates of non-target species such as duckweed (*Lemna gibba)* (Gardner and Grue 1996).

4.8.4 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Invertebrates

General studies on the impacts to invertebrates in wetlands of Puget Sound found that increased levels of toxic contaminants and changes in the water regime resulted in declines in taxa richness among the scraper and shredder functional feeding groups and the Chironomidae family (small, mosquito-like flies) (Ludwa and Richter 2001). These authors found declines in richness and abundance of invertebrate groups whose presence is seen as an indicator of the general health or quality of a water body. A second study in the Pacific Northwest also showed a direct and negative correlation between urbanization and the abundance and richness of macroinvertebrates (Hicks 1995) primarily through impacts to water quality.

The following sections first review the effects of metals on invertebrates and then describe the effects of organic and synthetic compounds such as pesticides. Much of the information on the impacts on invertebrates is based on studies in streams. These studies are applicable to wetlands because the species and groups of species reported in the studies are also found in wetlands.

4.8.4.1 Effects of Heavy Metals on Invertebrates

Heavy metals such as mercury, lead, zinc, copper, and cadmium can be directly toxic to wetland invertebrates. Metals can also impact invertebrate communities by altering the species composition and abundance of algae and aquatic plants upon which invertebrates depend for food and shelter. Growth, larval development, and reproduction of invertebrates can also be harmed by long-term exposure to sublethal concentrations of trace metals (Timmermans 1993). Relatively little, however, is known about the sublethal effects of metal pollutants in freshwater wetlands or how metals are metabolized or accumulated.

The extent to which heavy metals are toxic to wetland invertebrates depends largely on the acidity of the wetland and the particular form of the metal involved. Acidic conditions can mobilize and increase the toxicity of some metals, such as cadmium (Wright and Welbourn 1994), and decrease the toxicity of others, such as aluminum (Wren and Stephenson 1991). On the other hand, some metals, such as iron and aluminum, can to some degree protect invertebrates from otherwise toxic effects of heavy metals in acid mine drainage (Whipple and Dunson 1992).

Specific studies documenting the impact of heavy metals on invertebrates are summarized below:

- More than 20 years after cadmium and cobalt discharges to a freshwater marsh in New York were curtailed, invertebrate richness remained lower than at a control (less polluted) site (Klerks and Levinton 1993).

- Moderate recovery of invertebrates from metal contamination was demonstrated in the Coeur D'Alene River in Idaho. Over 22 years after contamination by zinc and other metals ceased, the number of taxa grew from zero to 18, while the proportion of mayflies, stoneflies, and caddisflies relative to the proportion of midges rose (Hoiland and Rabe 1992, Hoiland et al. 1994).

- Some studies show herbivores and detritivores as the most sensitive to additions of metals (Kiffney and Clements 1994a, Leland et al. 1989), whereas others have reported scrapers being the most sensitive group (Clements 1994).
- Mayflies and some stoneflies of western streams are sensitive to metals, whereas caddisflies and midges are relatively tolerant (Clements 1994, Kiffney and Clements 1994b, Leland et al. 1989, Nelson and Roline 1996).
- Agricultural drainage water containing arsenic, boron, lithium, and molybdenum entering the Stillwater Wildlife Management Area in Nevada proved acutely toxic to many wetland invertebrates (Hoffman 1992, Hallock and Hallock 1993a, 1993b).
- Copper and some other heavy metals appear to be more damaging to aquatic communities in the spring and summer rather than in the fall (Leland et al. 1989). Summer exposure to metals may coincide more closely with hatching of many macroinvertebrates, and early periods in the development of the invertebrates may be more susceptible.

4.8.4.2 Effects of Pesticides, Oil, and Other Contaminants on Invertebrates

Pesticides, oil, and other toxic contaminants represent a wide range of pollutants. In general, however, most have been shown to change the community structure (abundance, distribution, and richness) of invertebrates. Contaminants cause these effects through several mechanisms, including:

- Causing acute or chronic toxicity to invertebrates
- Altering algal communities and aquatic plants upon which some invertebrates depend for food and shelter
- Altering predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians
- Reducing rates of oxygen diffusion
- Changing the effects of other potential disturbances, such as acidity

The range of pesticides and organic pollutants used today is very large and it is not possible to generalize the impacts of this group of pollutants on invertebrates. Table 4-1 summarizes numerous studies that demonstrate the wide range of responses to contaminants by invertebrates.

Table 4-1. Summary of studies on effects of contaminants on invertebrates.

Reference	Contaminant Studied	Results
Eisler (1992)	diflubenzuron (insecticide)	In laboratory tests diflubenzuron was most toxic to crustaceans, followed by mayflies, midges, caddisflies. Larvae of corixids, dragonfly adults and larvae, spiders, dytiscids, and ostracods had moderate sensitivity
Eisler (1992)	paraquat, cyanide, fenvalerate, acrolein	These substances were lethal to invertebrates
Dieter et al. (1996)	phorate (pesticide)	In Prairie Pothole Region, macroinvertebrates that were particularly sensitive to phorate included hemipterans, mosquitoes, flies, mayflies, water mites, and water beetles. Less sensitive were leeches, snails, aquatic worms, ostracods
Lieffers (1990)	3-trifluoromethyl-4-nitrophenol (TFM) (lampricide)	TFM had a significant effect on invertebrates in a small stream
Fairchild and Eidt (1993)	fenithrothion (insecticide for forest insects)	Fenithrothion reduced emergence of aquatic insects for 6 to 12 weeks. Densities of most invertebrates (especially predatory species, midges, some other dipterans) were reduced by as much as 50% for more than one month after treatment. Wetland sediments became dominated by aquatic worms and water mites. Although in many streams and large lakes fenithrothion has transitory effects, residual toxicity remained in bog wetlands during winter and into the next year
Hachmoller et al. (1991)	various organic pollutants	Mayflies, stoneflies, caddisflies decreased in abundance in stream contaminated by various organic pollutants
Keller (1993), Metcalfe and Chaarlton (1990)	various contaminants	Mussels are especially sensitive to combined effects of pesticides, organic compounds, excessive nutrients
Kemp and Spotila (1996)	industrial pollutants, PCBs	Isopods, oligochaetes, craneflies were main survivors in a Pennsylvania stream with industrial pollution (including PCBs) compared with non-urbanized control segments
Crunkilton and Duchrow (1990)	oil	After 25 days, an oil spill in a Missouri stream reduced macroinvertebrate population to less than 0.1% of normal densities. Recovery of some species of stoneflies, mayflies, and caddisflies did not occur for at least nine months
Henry et al. (1994)	surfactant	In laboratory tests, a surfactant was approximately 100 times more toxic than the herbicide glysophate, with which it is commonly applied
Wipfli and Merritt (1994), Kreutzweiser et al. (1994a), Jackson et al. (1994), Waalwijk et al. (1992)	*Bacillus thuringiensis* var. *israelensis* (Bti) (biological control agent)	Bti appears to have minimal adverse effects on non-target insects in streams although mortality has been observed in Lepidoptera, some midges, craneflies, caddisflies, mayflies

Reference	Contaminant Studied	Results
Euliss and Mushet (1999)	agricultural contaminants	Direct adverse correlation found between aquatic invertebrate species richness and agricultural practices for seasonally inundated wetlands in prairie pothole region of North Dakota. Adverse effects on invertebrates could result from agrichemicals (shown to cause increased mortality of aquatic invertebrates in other studies). Tilling around wetland could increase erosion, leading to suspended sediments and adsorbed metals that are toxic to some zooplankton and thus affect the food chain

4.8.5 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Amphibians and Reptiles

Studies of the effects of heavy metals, pesticides, and other toxins on amphibians and reptiles have been conducted mainly on species, not communities. A review of relevant literature was published by Sparling et al. (2000). Schuytema and Nebeker (1996) have compiled a database of toxicity information from published literature for 58 amphibian species as related to 135 chemicals.

Many different pollutants have been documented as toxic to species of amphibians and reptiles found in Washington's wetlands. The following references document the impact of toxic compounds on some species found in the Pacific Northwest:

- Toxic effects of aluminum and other metals on the embryos and tadpoles of the northern leopard frog (*Rana pipiens*) were found by Freda (1989, 1991), Freda and McDonald (1990), and Freda et al. (1990).
- Many synthetic organic compounds affect amphibians and aquatic reptiles. Northwestern salamander (*Ambystoma gracile*) egg mortality corresponded with levels of total petroleum hydrocarbons in western Washington (Platin 1994, Platin and Richter 1995).
- The pesticide esfenvalerate caused damaging sublethal effects on tadpoles of the northern leopard frog (Materna et al. 1995).
- Tests of three forest insecticides (fenitrothion, triclopyr, and hexazinone) on the northern leopard frog in Ontario suggested that tadpoles were sensitive to triclopyr and fenitrothion (Berrill et al. 1991).

4.8.6 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Fish

The response of fish communities and individual species to toxic compounds is varied and complicated by many environmental factors. Smaller fish may be the first to respond to contaminants (Matuszek et al. 1990).

The toxicity of copper and zinc to some fish species depends on other chemical characteristics of the water (Munkittrick and Dixon 1992, Welsh et al. 1993, Erickson et al. 1996), as well as fish behavior (Pourang 1995). For example, dissolved organic matter from a marsh at a level of 5 mg carbon per liter kept copper from binding to the gills of small steelhead (*Oncorhynchus mykiss*), thereby reducing its toxicity. This occurred because copper formed a complex with dissolved organic carbon, making the copper unavailable (Hollis et al. 1997). In addition, some fish species may acclimate to moderately elevated levels of some metals (Klerks and Lentz 1998).

Selenium is not directly toxic to fish at usual concentrations but can become toxic once concentrated in fish food chains. This is especially true in some wetlands that receive effluents from irrigated fields or power plant reservoirs in some regions (Zilberman 1991, Lemly 1996).

Synthetic organics, including pesticides, can accumulate in wetland fish (Cooper 1991), often with adverse effects. In a Canadian wetland receiving effluent containing oily sand, fish had altered blood chemistry and died within 14 days (Bendell-Young et al. 2000).

4.8.7 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Birds

The response of individual bird species and bird communities to toxic compounds is varied. Individual species are directly affected by many pollutants. Many pesticides, however, are more likely to impact bird populations by altering their habitat and foods rather than by direct toxicity. Studies that document such impacts are summarized below:

- Several instances have been documented of wetland birds being directly poisoned by insecticides applied at recommended rates (e.g., parathion, as documented by Flickinger et al. 1991).
- Herbicides have been applied to wetlands to change the structure of vegetation and the species composition, with consequent shifts in the composition of bird species (Solberg and Higgins 1993, Linz et al. 1996). Information on pesticides in prairie wetlands has been compiled by Facemire (1992).
- Evidence of bird toxicity from lead shot used in hunting has been reported by Havera et al. (1992), Hohman et al. (1993), and Peters and Afton (1993).
- Detrimental reproductive effects from dioxins have been documented for great blue herons (*Ardea herodias*) (Hart et al. 1991); for dioxins and furans on wood ducks (*Aix sponsa*) (White and Seginack 1994); for PCBs (polychlorinated biphenyls) in American kestrels (*Falco sparverius*); and for petroleum in mallards (*Anas platyrhynchos*) (Holmes and Cavannaugh 1990).
- Research has continued to focus on the effects of selenium on waterfowl in western states. Biogeochemical conditions favoring the release of selenium into

wetlands are found throughout the arid regions of the western states and threaten bird communities in many wetlands along the Pacific and Central Flyways (Paveglio et al. 1992). Agricultural drainage, irrigation, and natural waters can leach selenium from many western soils. Subsurface irrigation is the most widespread and biologically important source of selenium toxicity for waterfowl, including the waterfowl in six national refuges (Ohlendorf et al. 1990, Feltz et al. 1991). Selenium is often accompanied by boron, which is toxic to ducklings (Stanley et al. 1996).

Impacts of lead shot

Lead in the aquatic environment can have significant impacts. Lead is toxic to aquatic biota (Eisler 1988). Waterborne lead is the most toxic form. Waterborne concentrations over 10 micrograms per liter have significant long-term effects on fish (Demayo et al. 1982). The introduction of lead into the aquatic food chain via aquatic plants has been found in the roots and foliage of the pond weed *Potamogeton foliosus* and in the exoskeleton of crayfish (Eisler 1988, Knowlton et al. 1983). Elemental lead (lead shot) has been shown to be significantly less bioavailable to rooted aquatics than powdered lead (Behan et al. 1979).

Waterfowl are at risk from ingesting lead shot as they forage in wetlands. Because of the proximity of wetlands to shooting ranges, other aquatic organisms, including amphibians, and some bird species may be at risk from the spent lead. For example, Eisler (1988) found that lead in tadpoles might contribute to the lead levels reported in wildlife that eats tadpoles. Predatory animals that feed on amphibians include reptiles (such as the garter snake), birds such as the great blue heron and red tailed-hawk, and mammals such as raccoons and coyotes (Meehan Martin, personal communication). The cleared range areas also encourage the introduction of rodent populations, which are preyed upon by the same predatory animals listed above.

Herbivorous land snails have been found to play an important role in cycling of lead in contaminated ecosystems (Dallinger and Wieser 1984, Beeby 1985). Therefore, snails and slugs in the forest ecosystem that graze in the gun range area may cycle lead into the forest food chain.

Plants growing in soils of low pH or low organic content readily accumulate lead (Demayo et al. 1982). Application of lime or phosphate, however, converts lead to hydroxides, carbonates, or phosphates of low solubility and reduces uptake by plants (Demayo et al. 1982). This in turn would reduce the amount of lead introduced into the food chain by herbivores.

4.8.8 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Mammals

Possible impacts of toxicity from pollutants on wetland mammals were not documented in the literature.

4.8.9 Summary of Key Points

- No information was found on the impacts of contaminants on the hydrologic functions of wetlands.
- The rates at which wetlands remove toxic compounds may actually be improved under low levels of contamination because the specific microbes that detoxify the pollutants are stimulated.
- The impact of contaminants on plants has not been studied as extensively, but the information suggests that toxicity from contaminants can change the composition of the plant community.
- Impacts of increased contaminants on the habitat provided by wetlands have been documented for invertebrates, amphibians, fish, and birds. Many contaminants are toxic to these species and their presence in wetlands reduces the suitability of a wetland as habitat. Mammals are the only group of vertebrates for which no information exists in wetlands.

4.9 Impacts of Changing Acidity

4.9.1 Impacts of Changing Acidity on Hydrologic Functions

No information was found on the impacts that increasing acidity might have on the hydrologic functions performed by wetlands. In the absence of any information to the contrary, however, it is possible to hypothesize that decreasing pH will probably not change how wetlands perform these functions. Changes in the acidity of water are not expected to change how well wetlands store water, how well they slow it down during peak flows, or how well they recharge groundwater.

4.9.2 Impacts of Changing Acidity on Functions that Improve Water Quality

Increased acidity (reduced pH) could change aspects of wetland chemistry that affect the ability to improve water quality. The rate of nitrification will probably decrease because, as noted by Mitch and Gosselink (2000), low pH inhibits denitrifying bacteria. The change in pH will also probably change the ability of the wetland to bind different toxic metals and other compounds.

No other information was found on the impacts that increasing acidity might have on how well wetlands remove pollutants.

4.9.3 Impacts of Changing Acidity on Plants

The pH is critical in determining the distribution of plants in wetlands. Changes in pH that result from human activities can, therefore, have major impacts. Studies described below have documented changes in plant populations that resulted from both decreases in pH (more acidic conditions) and increases in pH (less acidic conditions). However, the effects of acidification (or its reversal by liming) on the species composition of plants are not consistent among wetland types or even within individual wetlands (Farmer 1990, Baker and Christensen 1990, Mackun et al. 1994, Weiher et al. 1994).

For example, many plant species that inhabit bogs are adapted to acidity levels that would kill most wetland plants. Species whose decline or disappearance from a lacustrine wetland coincided with acidification include water lobelia (*Lobelia dortmanna),* shore quillwort (*Isoetes riparia*), water milfoil (*Myriophyllum tenellum)*, yellow pond lily (*Nuphar* sp.), common bladderwort (*Utricularia vulgaris)*, and ribbon leaf pondweed (*Potamogeton epihydris)* (Farmer 1990). Species whose relative abundance increased included *Leptodictium riparium,* needle spike rush (*Eleocharis acicularis),* sphagnum moss *(Sphagnum* sp.), and pipe wort (*Eriocaulon septangulare)* (Farmer 1990).

In general, making wetlands more acidic can directly impact plants by limiting the availability of some inorganic nutrients and carbon (Farmer 1990). Acidic conditions also promote the conversion of nitrates into ammonium.

Acidic conditions can impact plants indirectly by reducing the densities of invertebrates that graze or process detritus. Acidic conditions in wetland soils increase the toxicity of aluminum and manganese (Rendig and Taylor 1989, Crowder and Painter 1991).

4.9.4 Impacts of Changing Acidity on Habitat for Invertebrates

In general, changing the acidity in a wetland can alter the community structure of invertebrates by:

- Causing acute or chronic damage to tissues of invertebrates; species that easily lose sodium ions when pH is reduced tend to be most sensitive (Steinberg and Wright 1992)
- Altering algal communities and aquatic plants upon which some invertebrates depend for food and shelter (see discussion in Section 4.9.3)

- Altering the populations that are predators of invertebrates such as other crustaceans, amphibian, and fish (see Sections 4.9.5, 4.9.6)

The impacts of acidification on aquatic invertebrate communities have been researched extensively. Much of the information from Europe is compiled by Johnson et al. (1993). Table 4-2 categorizes invertebrate species as more or less tolerant of acidification based mainly on the North American literature. The list is included here because many of these species are probably found in Washington's wetlands. Few local studies, however, document the distribution of invertebrates in the state so it is not possible to identify the tolerance of species that are found here.

Some invertebrates are sensitive to pH increases (decreased acidity). For example, stormwater input to a Florida freshwater marsh increased phosphorus levels, lowered oxygen levels, and raised pH and hardness. This resulted in a shift of the macroinvertebrate population toward species that otherwise are intolerant of the acidic, nutrient-poor conditions typically found in the studied wetland (Graves et al. 1998).

Acidity often reduces the richness of macroinvertebrates in aquatic habitats (Schell and Kerekes 1989, Hall 1994). Another study showed that with increased acidity, many aquatic invertebrates declined in numbers and biomass, especially in wetlands with pH below 5.0 (Parker and Wright 1992). Reductions in acid emissions from some Canadian smelters was followed by significant increases in richness of invertebrates in water bodies downwind of the smelters (Griffiths and Keller 1992).

Table 4-2. Summary of studies describing relative tolerance of invertebrates to acidification.

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
Dragonflies and Damselflies (Odonata)		
Damselflies (Parker and Wright 1992, Baker and Christensen 1990)	X	
Some Odonata (*Enallagma civile*) (Giberson and MacKay 1991)		X
Beetles (Coleoptera)		
Some water beetles (Parker and Wright 1992), especially hydrophilid and dystiscid beetles (Baker and Christensen 1990)	X	
True Bugs (Hemiptera, Homoptera)		
Some water bugs, at least Notonectidae, Gerridae, Corixidae (Baker and Christensen 1990)	X	
Some water bugs (Parker and Wright. 1992)		X
Caddisflies (Trichoptera)		
Some caddisflies: *Cheumatopsyche pettiti* (Camargo and Ward 1992).	X	

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
Some caddisflies (Parker and Wright 1992) and some in the scraper and predator guilds (Williams 1991)		X
Flies, Midges, Mosquitoes (Diptera)		
Midges (Havens 1994a, Baker and Christensen 1990, Tuchman 1993), blackflies (Baker and Christensen 1989)	X	
Some midges, such as *Tanytarsus, Microtendipes*, and *Nilothauma* (Griffiths 1992)		X
Stoneflies (Plecoptera)		
Some stoneflies (Tuchman 1993) such as *Amphinemura* and *Leuctra* (Griffith et al. 1995)	X	
Many stoneflies, e.g., *Peltoperla arcuata* (Griffith et al. 1995)		X
Mayflies (Ephemeroptera)		
The mayfly *Eurylophella funeralis* (Griffith et al. 1995)	X	
Some mayflies (Balding 1992)		X
Other Macroinvertebrates		
Planarian *Dugesia dorotocephala* (Camargo and Ward 1992)		X
Some water mites (Havens 1994a)	X	
Molluscs (Grapentine and Rosenberg 1992, Gibbons and Mackie 1991, Balding 1992), including clams (Schell and Kerekes1989)		X
Mussels, snails, leeches (pH >5.0, Schell and Kerekes 1989)		X
The amphipod *Hyalella azteca* (Havens 1994a); pH must remain above 5.8 (Grapentine and Rosenberg 1992)		X
The amphipod *Gammarus minus* (Griffith et al. 1995)		X
Zooplankton		
Some zooplankters, such as *Daphnia galeata mendotae*, *D. retrocurva, Skistodiaptomus oregonensis* (Havens 1993)	X	
The rotifers *Gastropus stylifer, Keratella taurocephala, Polyarthra renata, Symchaeta* sp. (Fore et al. 1996)	X	
The water flea *Bosmina longirostris* (Havens 1993)		X
The rotifers *Asplanchna priodonta, Collotheca mutabilis, Conochiloides* sp., *Conochilus unicornis, Gastropus hyptopus, Kellicota longispina, Keratella*		X

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
cochlearis, Keratella crassa, Polyarthra dolichoptera, Trichocera cylindrica (Fore et al. 1996)		
Functional Feeding Groups		
Scrapers and collectors (Smith et al. 1990)	X	
Shredders (Tuchman 1993)		X
Deposit feeders (Smith et al. 1990)		X

4.9.5 Impacts of Changing Acidity on Habitat for Amphibians and Reptiles

Excessive acidity damages amphibians directly (Horne and Dunson 1994b). Acidity may also have direct impacts as a result of its capacity to mobilize toxic metals and perhaps by making sodium less available in some soil types (Wyman and Jancola 1991).

No studies were found describing the impact of increased acidity on amphibians and reptiles in Washington. Studies from other states, however, document these impacts. The information below summarizes some of the information for amphibian and reptile species that are found in the state, even if the studies were done elsewhere.

In Ontario, the acid-neutralizing capacity (alkalinity) of 38 wetlands positively influenced the probability of the northern leopard frog (*Rana pipiens*) being present (Glooschenko et al. 1992).

Embryos of the tiger salamander (*Ambystoma tigrinum*) had more than 70% survival at pH 4.5 and above but suffered much greater mortality at lower pH levels (Whiteman et al. 1995).

Concerns have been raised regarding the vulnerability to acidification of Montane wetlands in the West. Acidification makes aluminum and cadmium more mobile and increases their concentration in surface waters. Amphibians (e.g., Jefferson's and spotted salamanders) are known to be sensitive to acidity and elevated concentrations of aluminum found in some acidic ponds (Blancher 1991, Ireland 1991, Horne and Dunson 1995).

Aluminum released into Montane pools as a result of acidification sometimes has harmed embryos, reduced growth rates, and/or caused deformities and premature hatching of native amphibians (Bradford et al. 1991, Corn and Vertucci 1992).

4.9.6 Impacts of Changing Acidity on Habitat for Fish

No information was found on the impacts of acidity on fish in Washington's wetlands. In their review of the literature, Adamus et al. (2001) found that acidity can be directly toxic to fish, inhibit reproductive maturation, inhibit spawning behavior, induce emigration, and alter food availability. Furthermore, in areas where acid rain may be a problem, the increase in acidity induces aluminum toxicity in fish in many lakes and wetlands (Keller and Crisman 1990). Surveys of literature on effects of acidification on fish in lakes (and therefore potentially in wetlands along lake fringes) are provided by Baker and Christensen (1990) and Minns et al. (1990).

4.9.7 Impacts of Changing Acidity on Habitat for Birds

Acidification of wetlands affects birds primarily because it reduces the availability of calcium, which is important for egg development; potentially increases the availability of toxic metals; and alters the species composition and abundance of aquatic insects, submersed plants, amphibians, and fish that are important foods for waterfowl (see previous discussions in Sections 4.9.3, 4.9.4, 4.9.5, 4.9.6).

Changes in the types of available food, especially those rich in calcium, can diminish egg shell thickness and generally reduce the reproductive success of waterbirds in wetlands (Sparling 1990, 1991, Blancher and McNicol 1991, St. Louis et al. 1990, Albers and Camardese 1993). Overall, calcium deficiency appears to affect birds in acidified wetlands more than metal toxicity (Albers and Camardese 1993). Breeding pairs of 15 waterfowl species were more abundant in Ontario wetlands with over 40 parts per million (ppm) total alkalinity than in less alkaline wetlands (Dennis et al. 1989, Merendino et al. 1992). In British Columbia as well, densities of several breeding duck species were greater in ponds with higher levels of conductivity and calcium (Savard et al. 1994).

4.9.8 Impacts of Changing the Acidity on Habitat for Mammals

No information on the effects of acidification on the overall community structure of wetland mammals was located. It can be hypothesized, however, that where acidification becomes severe, community composition may shift from fish-eating species (e.g., otter, mink) to vegetarian or invertebrate-eating species and opportunists (e.g., muskrat, opossum) (Adamus and Brandt 1990).

4.9.9 Summary of Key Points

- No information was found on the impacts of acidity on the hydrologic functions of wetlands, but it is possible to hypothesize that impacts, if any, are minor.
- The rates at which wetlands remove toxic compounds are impacted by increasing acidity because the rates at which denitrification occurs are reduced.
- Increasing the acidity in wetlands can also change the composition of the plant community.

- Impacts of increasing acidity on the habitat provided by wetlands have been documented for invertebrates, amphibians, fish, and birds. In general, increased acidity reduces the richness of invertebrates in wetlands and impacts amphibians either directly or by changing the chemistry of the water in the wetland, making it less suitable as a habitat. Acidic wetlands also become less suitable habitat for birds because the amounts of calcium rich foods are reduced. Mammals are the only group of vertebrates for which no information exists.

4.10 Impacts of Increasing the Concentrations of Salt

Salt concentration in wetlands can increase as a result of (from Adamus et al. 2001):

- Isolating wetlands from some types of groundwater inflow
- Increasing water lost through evaporation
- Discharging effluents (especially irrigation return water)
- Routing runoff that has relatively high conductivity into wetlands

Increased concentrations of salt (salinization) impact the functions of wetlands as described below.

4.10.1 Impacts of Increasing Concentrations of Salt on Hydrologic Functions

No information was found on how changes in salt content might affect the hydrologic functions of flood storage and flood desynchronization. In the absence of any information to the contrary, however, it is possible to hypothesize that salinization will probably not change how wetlands perform these functions. Changes in the salt content of water are not expected to change how well wetlands store water, how well they slow it down during peak flows, or how well they recharge groundwater.

4.10.2 Impacts of Increasing Concentrations of Salt on Functions that Improve Water Quality

One relevant study found that salinities greater than about 300 grams per liter can inhibit the ability of microbes to detoxify toxic forms of selenium (Steinberg and Oremland 1990). This was the only literature found on how salinization might impact the ability of wetlands to remove pollutants.

As noted below, salinization has some impacts on plants, and thus it may affect nutrient uptake and transformation in a wetland. However, it is not possible to predict or hypothesize how such changes in these species might change other functions that improve water quality.

4.10.3 Impacts of Increasing Concentrations of Salt on Plants

In general, high concentrations of soluble salts are lethal to freshwater plants, and lower concentrations may impair growth (Rendig and Taylor 1989). Woody plants tend to be less tolerant than herbaceous plants because they do not have mechanisms for removing salt, other than accumulating salts in leaves and subsequently dropping them (Adamus et al. 2001).

Many plant species that inhabit inland saline wetlands are, of course, adapted to tolerating salt levels that would kill most other wetland plant species. A survey of inland lakes in western Canada which spanned a salinity gradient identified relative tolerance to salinity and specific salinity tolerance thresholds of many wetland species (Hammer and Heseltine 1988).

Individual plant species have different tolerances and reactions to increasing salinity. It can be expected that the plant community in a wetland will change to one dominated by salt-tolerant plants when additional salts are introduced. For example, wetlands in which salt has been present for some time, such as alkali wetlands, have a completely different plant community than that found in non-alkali wetlands. In eastern Washington a major change in plant communities was found when the conductivity (a measure of the amount of salts present in the water) increased to 2.0 milliSiemens and higher (Hruby et al. 2000).

A study by Hutchinson (1991) describes the tolerance of many wetland plants found in Washington. It can be used to predict how the plant species might change in Washington's wetlands as salt concentrations increase.

It can also be expected that wetlands subject to increases in salinity through agricultural practices or discharges of salt will also be subject to a change in plant populations. One wetland undergoing such a change was observed in the Richland area during the calibration of the Washington State Wetland Rating System for Eastern Washington in the summer of 2002. The conductivity of the wetland was measured at about 6.5 milliSiemens. About one-quarter of the area was still dominated by cattails (*Typha latifolia*), a wetland plant with a relatively low tolerance to salt (Hutchinson 1991), but this species was dying. Dead stalks of this species covered almost half the area of the wetland.

4.10.4 Impacts of Increasing Concentrations of Salt on Habitat for Invertebrates

The review of the literature indicates that high levels of salinity can alter the structure of freshwater invertebrate communities in many ways. Adamus et al. (2001) have identified the following mechanisms by which the invertebrate community can be altered:

- Acute and chronic damage to tissues of invertebrates

- Changes in the species composition and structure of algal communities and aquatic plants upon which some invertebrates depend for food and shelter
- Changes in predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians
- Changes in the bioavailability of some other substances, such as heavy metals and nutrients

Even at low concentrations, increases in chloride (a correlate of salinity, and often associated with road salt applications) among 27 Minnesota wetlands were significantly correlated with declines in species richness among the wetlands (Gernes and Helgen 1999). In Wyoming wetlands of fairly low salinity (0.8 to 30 milliSiemens per centimeter), the dominant macroinvertebrates were amphipods and epiphytic snails. Other recent species-specific salinity data for wetland invertebrates are presented Parker and Wright (1992), and Lovvorn et al. (1999).

4.10.5 Impacts of Increasing Concentrations of Salt on Habitat for Amphibians and Reptiles

In general, relatively little is known about amphibian tolerance to salinity. Three studies have reported a statistically significant negative correlation between conductivity of the water and amphibian species richness (Azous 1991, Platin 1994, Platin and Richter 1995). However, the implications of these studies for understanding impacts on existing populations of amphibians in a wetland that is undergoing an increase in salt concentrations is not clear.

4.10.6 Impacts of Increasing Concentrations of Salt on Habitat for Fish

No information was found on the tolerance of native fishes in Washington to salinity. Adamus et al. (2001) reported the following information relative to some of the introduced game fish that now are found in Washington's wetlands.

Laboratory trials consisting of 120-day exposure of freshwater largemouth bass (*Micropterus salmoides*) to four salinity levels (0, 4, 8, and 12 ppm) indicated a significant decrease in growth rate with increasing salinity up to 8 ppm.

In another experiment, juvenile bluegill (*Lepomis macrochirus*) from a freshwater pond in northeastern Mississippi and a brackish bayou in coastal Mississippi were held in a chamber with zero salinity but given access to chambers containing 0, 2, 4, 6, 8, and 10 ppm salinity (Peterson et al. 1993). Fish from neither habitat showed a clear preference for any of the salinity options. These data and data from previous studies suggest bluegills are better able to physiologically and behaviorally tolerate elevated salinity relative to other centrarchids (the family of fish containing bluegills, bass, crappies, etc.), particularly bass (Peterson et al. 1993).

4.10.7 Impacts of Increasing Concentrations of Salt on Habitat for Birds

The impacts of increasing salinity on birds are highly dependent on the species in question. The following summarizes relevant studies:

- Highly saline or alkali conditions are detrimental to some invertebrate and plant foods used by many duck species. High salinity is directly toxic or impairs the growth of young ducklings (Clark and Nudds 1991, Moorman et al. 1991).
- Sensitive waterbirds, such as some ducks, may visit saline wetlands but often only when fresher wetlands are available nearby (Lokemoen and Woodward 1992, Woodin 1994, Adair et al. 1996).
- Breeding densities of most duck and grebe species in interior British Columbia were greater in ponds with higher conductivity, but marsh nesting species were unaffected (Savard et al. 1994).

Nonetheless, a few species of water-birds occur regularly at very high densities in alkali wetlands during the breeding season and/or migration. Examples include the American avocet (*Recurvirostra americana*), snowy plover (*Charadrius alexandrinus*), phalaropes, killdeer (*Charadrius vociferus*), horned grebe (*Podiceps auritus*), tundra swan (*Cygnus columbianus*), and white-rumped, semipalmated, and Baird's sandpipers (*Calidris* spp.) (Earnst 1994, Jehl 1994, Savard et al. 1994, Oring and Reed 1997, Rubega and Robinson 1997, Warnock 1997). These relatively salt-tolerant species also occur in less saline wetlands, but their abundance often is greatest in wetlands with high salinity, and is related to sharp seasonal peaks in the abundance of brine shrimp and other salt-tolerant invertebrates. These birds characteristically travel hundreds of miles, sometimes daily or weekly, in order to exploit such invertebrate foods during the short times when the food peaks (Haig et al. 1997).

4.10.8 Impacts of Increasing Concentrations of Salt on Habitat for Mammals

No information was located on the impacts of salinization on the overall structure of mammal communities in wetlands and the suitability of wetlands as habitat for mammals.

4.10.9 Summary of Key Points

- No information was found on the impacts of salinization on the hydrologic functions of wetlands, but it is possible to hypothesize that impacts, if any, are minor.
- Only one study was found that documents any impacts of salinization on the ability of wetlands to improve water quality. Very high salt concentrations inhibit the microbes that detoxify selenium.
- Increasing the salt concentrations in wetlands can change the composition of the plant community.
- Impacts of increased salt concentrations on the habitat provided by wetlands have been documented for invertebrates, fish, and birds. In general, increased salinity changes the composition of the invertebrate community in wetlands. Largemouth bass seem to be especially sensitive to increased salinity relative to other species. The ducklings of some waterbird species may also be sensitive. No information exists on the impact of salinization on mammals and amphibians.

4.11 Impacts of Decreasing the Connection Between Habitats

Decreasing connections between habitats (fragmentation) results directly from human conversion of land to uses that are not part of an undisturbed ecosystem. Fragmentation is a result of both the direct loss of wetlands that isolates populations of wildlife and the creation of barriers to the movement of organisms. Wetland loss and isolation is seen as a major factor contributing to the loss of biological diversity in vertebrate populations that use wetlands (Harris 1988, Gibbs 2000). In general, fragmentation of habitats affects biological diversity through (Harris 1988):

- Loss of the species less tolerant to disturbance or those that inhabit the interior parts of wetlands
- Loss of large species with broad ranges
- Loss of genetic integrity within populations
- Increase in numbers of habitat generalists that thrive in disturbed environments, such as parasites

Occasional migration between wetlands is vital in sustaining local populations of wetland-dependent organisms. Limiting the movements of these species reduces the exchange of genetic material among local populations and can result in population extinctions (Gibbs 2000). Three factors that impede movement among wetlands and other habitats include (Gibbs 2000):

- Greater distances between wetlands
- Degradation of upland habitats
- Increased road density

The effects of fragmentation on wildlife that use wetlands are most extensively documented for amphibians and birds. Little information is available for effects on macroinvertebrates, reptiles, and mammals. Several studies done in the Pacific Northwest are cited in the following discussion of how decreasing habitat connections affects wetland functions.

4.11.1 Impacts of Decreasing the Connection Between Habitats on Hydrologic Functions

Information on how fragmentation affects the flood storage, flood desynchronization, and groundwater recharge performed by individual wetlands was not located in the literature. In the absence of any information to the contrary, however, it is possible to hypothesize that fragmentation will probably not change how individual wetlands still remaining in the landscape perform these functions. Fragmentation at a landscape level is not expected to change how well the remaining individual wetlands store water or how well they slow it down during peak flows. On the other hand, fragmentation probably does impact the delivery and routing of water to wetlands as described in Chapter 3. This may change how much water gets to a wetland for storage but not how well the wetland can store it.

4.11.2 Impacts of Decreasing the Connection Between Habitats on Functions that Improve Water Quality

Information on how fragmentation affects the ability of wetlands to improve water quality was not located in the literature. It is not possible to predict or hypothesize precisely how such changes might affect these functions.

4.11.3 Impacts of Decreasing the Connection Between Habitats on Plants

No information on the response of plant communities to fragmentation was found.

4.11.4 Impacts of Decreasing the Connection Between Habitats on Invertebrates

Few studies were found that documented the impact of decreasing connections on the suitability of wetlands as habitat for invertebrates. One study found that wetland isolation combined with the harshness of the surrounding upland landscape in more arid environments (such as much of eastern Washington) limit dispersal and colonization by aquatic invertebrates (Myers and Resh 1999).

Another study in New York comparing macroinvertebrate populations at restored wetlands and reference wetlands showed that less mobile invertebrates colonized new wetland sites very slowly or not at all, whereas insects that disperse aerially colonized the new sites rapidly (Brown et al. 1997). Therefore, wetland isolation may have greater effects on less mobile invertebrate species.

4.11.5 Impacts of Decreasing the Connection Between Habitats on Amphibians and Reptiles

4.11.5.1 Amphibians

As early as the mid-1960s, researchers in various parts of the country perceived the effects of reduced connection of habitats on amphibians. One author notes the disappearance of a number of species of frogs, toads, turtles, and snakes in an urbanizing area in the Midwest that he studied from 1949 to 1964 (Minton 1968).

The effects of increased wetland isolation have been extensively studied for amphibians since then. This is probably because amphibians:

- Are restricted to movement on the ground
- Do not typically have large migration ranges
- Often move between terrestrial and aquatic habitats
- Have experienced significant population declines throughout the world

The causes of declines in the populations of amphibians have been extensively studied and most researchers conclude that the problem is very complex and multiple factors are likely at work (Hayes and Jennings 1986, Pechmann et al. 1991, Pechmann and Wilbur 1994, Delis et al. 1996, Adams 1999). Among these factors, there is evidence that increasing isolation of wetlands due to wetland loss may play a significant role in declining amphibian populations (Ostergaard 2000, Adams 1999, Lehtinen et al. 1999, Semlitsch and Bodie 1998). This has significant implications for amphibians in Washington State because about 57% of amphibian species that occur here commonly use wetlands for at least one life cycle stage (Leonard et al. 1993).

Amphibians are not randomly distributed within acceptable habitats—they occur in higher abundance and species richness in habitats that are better connected to other desirable habitats (Lehtinen et al. 1999, Lehtinen and Galtowitsch 2001). A Minnesota study of 21 marshes noted that the two most important predictors of decreases in amphibian species richness in agricultural areas are the degree of wetland isolation and the road density (Lehtinen et al. 1999). The marshes in this study were located in both prairie and hardwood forest ecoregions in two primary land use settings: urban and agricultural. The study noted some differences between ecoregions and land use effects. In the agricultural prairie ecoregion, the amphibian assemblages observed appeared to be most influenced by:

- Road density
- Wetland isolation
- Biological interactions (presence of predators)

In deciduous forest areas that are urbanizing, amphibian richness was most closely related to upland land use and associated habitat fragmentation.

Other landscape-based studies also conclude that the distances between wetlands, as well as the suitability of terrestrial habitats, are key factors in amphibian distribution. Amphibian recolonization patterns are species and spatially dependent because not all species have the capacity to move beyond fragmented, isolated habitats (Lehtinen and Galatowitsch 2001).

Declines in the richness of amphibian species have also been documented as urban land use increases (Lehtinen et al. 1999, Knutson et al. 1999, Richter and Azous 2001a). A landscape analysis of habitats for anurans (frogs and toads) in Wisconsin and Iowa showed that anurans were positively associated with uplands, wetland forests, and emergent wetlands and negatively associated with urban land (Knutson et al. 1999). A positive association, in this case, means higher abundance and species richness. The negative association with urban land is attributed by the authors to:

- Conversion of habitat
- Roads acting as barriers
- Presence of exotic predators
- Chemical contamination
- Other factors

A study of frog distribution in the Netherlands found that the likelihood of a pond being used by frogs depended on the density of ponds and the amount of suitable terrestrial habitat in the surrounding area (Vos and Stumpel 1995). A similar study in the Netherlands showed that frog use of ponds was negatively correlated with the degree of wetland isolation and road density in the surrounding landscape (Vos and Chardon 1998). Distances between breeding ponds and other life stage habitats, as well as the condition of the terrestrial habitats, were primary factors in determining frog distribution. Open fields were avoided by adults and newly metamorphosed juveniles. Roads increased the mortality of frogs and acted as barriers between wetlands, thus effectively increasing wetland isolation (Vos and Chardon 1998).

Similarly, an Indiana study concluded that amphibian distribution was influenced by (Kolozsvary and Swihart 1999):

- Forest area and proximity
- Density of ponds
- Degree of wetland permanency
- Density of vegetation

The importance of each factor varied for each species.

Using a simulation model, one author concluded that the amount of breeding habitat had a significantly greater effect on the likelihood of population extinction than the extent of habitat fragmentation (Fahrig 1997). Her model showed that if breeding habitat covers more than 20% of the landscape, population extinction is very unlikely no matter how fragmented the habitat. However, this work was based on a generalized model that made a number of assumptions that cannot be verified without targeting a selected species, as do the more empirical studies of amphibian distribution.

Other studies indicate that there is a threshold for extent of wetland isolation or distance between wetlands for each amphibian species. Several studies of maximum distances of amphibian movement to breeding habitats indicate that amphibian reproductive success is affected by wetland isolation and terrestrial habitat condition:

- Richter and Azous (1995) suggest that upland forest habitat must lie within 3,280 feet (1,000 m) of breeding wetland habitat for it to be useful to lentic (pond) breeding amphibian species.
- Baker and Halliday (1999) found limits on the distance that species of newts, frogs, and toads would move to colonize new ponds in England (1,312 feet [400 m] for newts, 3,117 feet [950 m] for frogs and toads). In contrast to other studies, the condition and nature of the adjacent upland habitats did not have a strong correlation to pond colonization. The study may not have been sensitive enough, or the mixed land uses within the agricultural settings may have actually supported amphibian populations.
- The ability of juveniles to move from one wetland to the next depends on the spacing between wetlands and the habitat conditions within the buffers. Distances between ponds directly affect the probability of recolonization and the chance to prevent extinction of amphibian populations. Most individual amphibians cannot migrate long distances and adults return to their home ponds, usually after migrating no more than 656 to 984 feet (200 to 300 m) (Semlitsch 2000).
- A similar study in the Netherlands showed that amphibians would colonize new ponds up to 3,280 feet (1,000 m) away (Laan and Verboom 1990). The authors concluded, however, that the probability of a species colonizing a wetland

increases with proximity to the source wetland and increased connectivity by upland forest habitats between the wetlands.

4.11.5.2 Reptiles

No studies were found that specifically addressed the effects of reduced habitat connections and wetland loss on reptiles. In one study in North Carolina, researchers evaluated the adequacy of federal and state wetland regulations in protecting the habitats that freshwater turtles need to complete their life cycles (Burke and Gibbons 1995). They determined that the area protected as wetland under federal guidelines did not include the area in which two critical life-cycle stages occurred: nesting and terrestrial hibernation. This means that some of the habitats needed for turtle success are vulnerable to loss due to conversion to other land uses. However, this study focused not on the effects of wetland loss but the effects of eliminating upland habitats adjacent to wetlands.

A study that modeled the effects of wetland loss in Maine showed that local populations of freshwater turtles faced a significant risk of extinction following the loss of small wetlands (Gibbs 1993).

As with amphibians, the limited dispersal distances of reptiles, in comparison to birds and mammals, would logically make reptiles particularly vulnerable to habitat fragmentation. However, documentation of the effects of habitat fragmentation on reptiles that use wetlands is very sparse. It appears to be completely lacking for Washington State. This may be due in part to the fact that, with the exception of the western pond turtle (*Clemmys marmorata*), a listed species in the state, no reptile species in Washington are primarily dependent on aquatic habitats. However, the western terrestrial and common garter snakes (*Thamnophis* spp.) are both common near water bodies, including wetlands.

4.11.6 Impacts of Decreasing the Connection Between Habitats on Fish

No information was found on the impacts of fragmentation on the suitability of wetlands as habitat for fish.

4.11.7 Impacts of Decreasing the Connection Between Habitats on Birds

The impacts of decreasing connections between habitats have generally been studied in two types of fragmented landscapes: one fragmented by growing urbanization and one fragmented by agricultural practices. In general there are no studies or conclusions in the literature that would suggest the fragmentation from these two types of land use has significantly different impacts on populations of birds, and so both types of studies are reported below.

The extent of wetland isolation is known to be an important factor that influences bird use of wetland habitats:

- In a study of Puget Sound wetlands, researchers documented a positive association between bird species richness and the proximity of lakes and open water habitats, as well as the structural complexity of the vegetation in the wetlands (Richter and Azous 2001b).
- In northern prairie marshes, bird species richness declined with increased isolation of the wetland (Brown and Dinsmore 1986). Marshes that were part of wetland complexes showed higher species richness than isolated wetlands. Smaller marshes had occurrences of certain bird species only when the marshes were part of a wetland complex.
- These findings are supported by a more recent study of wetland complexes in prairie marshes in Iowa (Fairbairn and Dinsmore 2001). This study related bird species richness and densities of individual species to habitat variables within the wetland complexes and to area of wetland habitat in the surrounding landscape. For some bird species, presence and abundance in a wetland complex were clearly related to the amount of wetland habitat in a 1.9 mile (3 km) area surrounding the complex. A similar study also determined that unfragmented landscapes with prairie marsh supported more waterfowl species than isolated wetlands (Naugle et al. 2001).

The pattern of wetland habitat use varies between different wetland-dependent bird species (Naugle et al. 1999):

- Some species are sedentary and rarely use resources beyond the nest vicinity
- Some use only larger wetlands regardless of the surrounding landscape
- Others require a mosaic of wetlands on the landscape

Therefore, the entire landscape must be assessed, rather than just the wetland patches, in order to determine the habitat suitability of an area for wide-ranging species.

A correlation has been found between the degree of urban development in an area (and the resultant fragmentation) and the extent of declines in native bird species richness. One study in Santa Clara County, California, looked at six sites representing a gradient of development ranging from biological preserve to business district (Blair 1996). Increasing proportions of invasive and exotic bird species were found in the more highly developed areas. The moderately developed sites were highest in species richness and bird biomass. They were, however, lower in native bird diversity than the lesser disturbed sites. The shift in species was related to changes in total available habitat and in habitat structure across the gradient. This study concluded that even relatively minor habitat alterations resulted in loss of species.

Wetlands in the Puget Sound area showed a similar response to urbanization. Researchers found no correlation between total bird species richness and amount of impervious surface, but there was a correlation with native species richness (Richter and Azous 2001b). The rarer, more sensitive birds, all of which are native, tended to decrease with urbanization. The more adaptive species, with a higher percentage of non-natives (e.g., European starlings [*Sturnus vulgaris*]), tended to increase in urbanizing watersheds. Again, these changes are most likely due to loss of habitat, and therefore reduced connections between habitats, as well as habitat degradation.

One study has important implications because it indicates that duck breeding and brood raising are most successful with a variety of wetlands in close proximity. Conducted in eastern Canada it examined the role that habitat heterogeneity plays in the use of wetlands by ducks (Patterson 1976). It concluded that breeding duck pairs spaced themselves based on the physical size of the wetland. The authors also observed that breeding can occur in relatively "sterile" wetlands (those with hard water). However, duck broods hatched in more sterile wetlands often moved to more biologically productive wetlands where there was a greater food source and more refuge/escape habitat. These preferable wetlands were close to the breeding wetlands because young waterfowl cannot fly.

As with amphibians, the presence of terrestrial habitats between wetlands can be an important factor in waterfowl distribution. A study conducted in an area of intensive wheat farming demonstrates the importance of maintaining connections among habitats for birds (Saunders and DeRebeira 1991). These researchers found that native bird species used corridors as narrow as 13 feet (4 m) to move between patches of preferred habitat. Corridor width was positively correlated with species richness.

A study of bird populations in forest interiors found that habitat fragmentation impairs reproduction and can result in population declines and extinctions (Temple and Cary 1988). Though not focused on wetlands, the study can reasonably be applied to forested wetlands. The authors modeled the effects of habitat fragmentation. They predicted that success rates for nests for forest-interior birds would drop from 70% when nests are greater than 656 feet (200 m) from the forest edge, to only 18% when nests are less than 328 feet (100 m) from the edge. This indicates that fragmentation of forested wetlands through such activities as logging could have significant effects on species that are not tolerant of edge habitats.

In Minnesota, Mensing et al. (1998) assessed the implications of fragmentation at various landscape scales for birds. They found that:

- Diversity and richness of bird species increased with an increase in the extent of forest and wetland within the surrounding landscape.
- Habitats that were in good condition in the areas surrounding wetlands strongly influenced the biotic diversity, with positive correlations shown for birds within 1,640 feet (500 m) of the wetland edge.

4.11.8 Impacts of Increasing Connections Between Habitats on Mammals

Information on the effects of wetland habitat loss and fragmentation on mammals is sparse, even though a number of the mammal species in Washington State are known to commonly use wetlands (beaver, muskrat, mink, otter, water vole, deer mouse, and others). Most of the literature addresses the effects of beaver dams on wetland systems.

One study from the Pacific Northwest documented that wetland fragmentation and the elimination of surrounding upland habitats can have significant effects on small mammals. Richter and Azous (2001c) found that the total area of undeveloped land adjacent to a wetland (including forest, shrub, agricultural fields, and meadows) was weakly associated with mammal richness. A stronger correlation was between the percent of adjacent forest land (within 1,640 feet [500 m] of a wetland) and mammal richness. The highest small-mammal richness was observed in wetlands with at least 60% of the first 1,640 feet (500 m) surrounding the wetland in forest. The authors noted that mammal species richness in Puget Sound wetlands has no correlation with area of impervious surface in the watershed.

Roads are an important factor in habitat fragmentation. For example, a major highway in Massachusetts increased wetland isolation and blocked major travel corridors between suitable habitat patches for mammals (Forman 1998). See Section 4.12.2 for additional discussion of effects of roads on wildlife.

4.11.9 Summary of Key Points

- No information was found on the impacts of fragmentation on the hydrologic functions or the functions that improve water quality.
- Increased wetland isolation appears to be a major factor in species richness and abundance for all taxonomic groups. One author states that "modifications to the environment that preclude movement between component subsystems may be as devastating to vertebrates in the long run as are forces that actually destroy the wetland" (Harris 1988).
- Impacts of fragmentation on the habitat provided by wetlands have been documented for invertebrates, amphibians, reptiles, birds, and mammals. No information was found on impacts to habitat for fish and on the distribution of plants in wetlands.
- Wetland complexes are important to amphibian success. The loss of connections between wetlands has caused reductions in both amphibian abundance and richness.

- The impacts of habitat fragmentation are not as well documented for birds and mammals as they are for amphibians. Certainly there are different issues and patterns of habitat use between these taxonomic groups.

4.12 Impacts of Other Human Disturbances

Human activities on the land create many different types of disturbances. The previous discussion addressed only the major ones that have been studied. The following sections review some of the impacts of other types of disturbances that have been documented to a lesser extent. The discussions in these sections are not separated by wetland function because all of the impacts address either plants or wildlife, and the information is not extensive enough to warrant subdividing it.

4.12.1 Impacts of Altering Soils on Plant Communities

Physically disturbing wetland soils during the dry season, through tillage, compaction, excavation, or other means, can allow invasion by non-native plant species (Morin et al. 1989, Sutton 1996, David 1999, Galatowitsch et al. 1999). It can also destroy much of the viable seed bank (Lee 1991). Tilling the soil often reduces diversity, including both richness and evenness, as documented in a Carolina bay wetland (Kirkman and Sharitz 1994). The tillage treatment disrupted the roots of perennials more than burning, and it encouraged germination of annuals in the seed bank and colonization by several invasive species.

Invasive plants, especially non-native plants, significantly alter the species composition of many wetlands, sometimes even forming nearly monotypic stands. Among the most widespread invaders in North America are cattail (*Typha*), reed canarygrass (*Phalaris* sp.), purple loosestrife (*Lythrum salicaria*), giant reed (*Phragmites* sp.), milfoil (*Myriophyllum spicatum*), and hydrilla (*Hydrilla verticillata*). Their increased dominance is frequently attributed in part to the physical disturbance of soils or water levels within a wetland and/or the surrounding landscape, including accelerated sedimentation, eutrophication, and the construction of mitigation wetlands (Confer and Niering 1992, Magee et al. 1999).

Continuously disturbing the soil, for example through compaction and road building, can alter species composition. These disturbed conditions can lead to a decline in both the biomass of native species and a change in the soil conditions that support them (Ehrenfeld and Schneider 1991). Use of all-terrain vehicles also impacted wetlands on the Atlantic coastal plain, reducing the density of seed in wetland seed banks and allowing common rushes to displace rare species (Wisheu and Keddy 1991). Excavation and clearing of gas pipeline rights-of-way through forested wetlands in Florida resulted in increased species richness within the wetland clearings but an increased percent cover of non-native species (van Dyke et al. 1993).

4.12.2 Impacts of Roads on Wildlife

Roads contribute to lower species richness for a variety of wildlife groups through the factors listed below. While all of the studies cited in this section were conducted in other regions of the country, much of the information is likely to pertain to effects on Pacific Northwest wildlife because the effects are inherent to roads regardless of region.

It is theorized that roads cause the loss of biodiversity by (Findlay and Bourdages 2000):

- Restricting movement between populations of wildlife
- Increasing mortality
- Fragmenting habitat
- Increasing edge habitat
- Facilitating invasion by exotic species
- Increasing human access to wildlife habitats

Findlay and Bourdages (2000) note that there may be long time lags between road construction and the time when effects on wildlife are perceptible. Effects may be undetectable in some taxa for decades.

Increased road density is implicated in lower species richness and abundance for vertebrates. In wetlands in southeast Ontario, the species richness of mammals, amphibians, reptiles, and birds was seen to decline with increased road density (Findlay and Houlahan 1997). Road construction and forest removal are viewed by these authors as increasing the risk of loss of biodiversity in wetlands.

Frog and toad density decreased with increasing traffic in another study by Fahrig et al. (1995). This study concluded that increased road density can contribute to amphibian population declines in urbanizing areas. A study of amphibians using small isolated wetlands in Florida found high mortality during migration between upland terrestrial habitats and temporary pond breeding habitats (Means 1996). The author attributes much of this to direct road mortality.

A study of the "road-effect zone" of a four-lane suburban highway in Massachusetts was undertaken to determine the distance from a road that impacts can be measured (Forman 1998). This study concluded that the road blocks migration routes for salamanders up to several hundred meters from wetlands. The study also showed that the effect of the road on blocking major travel corridors between suitable habitat patches for small mammals could be measured to several kilometers from the road. The effects of traffic noise on birds could be measured up to 2,132 feet (650 m) from the road in forested areas and 3,051 feet (930 m) in open areas. The implications of effects on wetland wildlife are evident even though the findings of this study are applicable to a variety of habitats. (See Section 4.12.3 for more on the effects of noise on wildlife.)

A related study of the same Massachusetts highway showed that significant ecological effects extended out at least 328 feet (100 m) from the highway. Forman and Deblinger (2000) studied nine ecological factors relating to, among other things, wetlands, streams, and amphibians. Assessing all factors, this study concluded that the “road-effect zone” averaged approximately 1,969 feet (600 m) wide, though it was quite variable in width at specific locations.

4.12.3 Impacts of Noise on Wildlife

The effect of noise on wildlife is a topic of growing concern. The frequency of the sound waves and the duration of the sounds influence how noise affects wildlife species. Although many of the studies discussed below do not address wetlands specifically, the impacts of noise are not expected to change whether the species in question is in a wetland or another type of habitat.

Frequency is the perceived pitch of sound, and different animals show different sensitivities to the same range of frequencies. Generally, smaller mammals such as rodents, shrews, and bats have a greater sensitivity to higher frequencies—often within ranges exceeding 20,000 Hertz (Hz), the upper limit of human sound perception. Larger mammals show sensitivity to low frequencies and may be able to detect sound at or below 10 Hz. While most birds show auditory sensitivity similar to humans (20 to 20,000 Hz), certain birds (such as rock doves) can also perceive low-frequency sounds, often with much greater sensitivity than their larger mammalian counterparts (Kreithen and Quine 1979). Some frogs and toads also show low-frequency sensitivity (Hetherington 1992), and even some small mammals are capable of discerning sounds of only a few Hertz (Plassman and Kadel 1991).

Sound duration may be divided into two classifications: continuous sounds which last for a long time with little or no interruption, and impulse sounds lasting for only short durations (Larkin et al. 1996). Impulse sound and continuous sound appear to have different physiological and behavioral effects. Generally, impulse noise appears to be more stressful to wildlife, at least in part due to the unpredictability of such noise (Larkin et al. 1996).

Overall, the literature suggests that species differ widely in their physiological response to various types, durations, and sources of noise (Manci et al. 1988). However, noise effects on wildlife may be broadly classified as primary, secondary, and tertiary:

- **Primary** effects are direct, physiological changes to the auditory system and may be considered to include the “masking” of auditory signals. Masking is the inability of an individual to hear important environmental signals such as calls from mates or noises of predators or prey.

- **Secondary** effects may include non-auditory physiological effects such as stress and hypertension, as well as behavioral modifications that include interference with mating or reproduction and impaired ability to obtain adequate food, cover, or water.

- **Tertiary** effects are the direct result of primary and secondary effects at a population level and include population decline and habitat degradation. Most of the effects of noise are mild enough that they may never be detectable as variables of change in population size or population growth against the background of normal variation (Bowles 1995).

The behavioral responses of wildlife to noise show a high degree of variation depending on the species, the type of noise, and the habituation of the individuals to the source of noise. For example, some bald eagles can be very tolerant of auditory stimuli when the sources are screened from view (Stalmaster 1987), but other raptor species such as prairie falcons flush from perches and nests at sudden loud noises (Harmata et al. 1978).

Animals may become tolerant of repeated noises. Krausman et al. (1986) studied desert ungulates exposed to aircraft noise and noted that short-term habituation to aircraft noise occurred with repeated exposure. Sandhill cranes nesting meters away from a Florida highway showed no response to passing traffic (Dwyer and Tanner 1992). The effects of noise vary not only with the type of noise in question, but with an individual animal's experience, time of day (Herbold et al. 1992, Gese et al. 1989), and reproductive cycle (Platt 1977).

Research on the effects of traffic noise on breeding birds was conducted by Reijnen et al. (1995, 1996) who studied woodland and grassland bird populations in the vicinity of roadways. Ambient noise up to a given level resulted in no reduction in the density of bird populations. However, once an ambient noise threshold level was exceeded, densities decreased exponentially with increased noise. Threshold levels were found to range from 36 to 58 decibels, depending upon species, and the zones of decreased breeding densities surrounding the roadways ranged up to 2,670 feet (810 m) for particularly sensitive species near busy roadways. They found habitat avoidance by individual birds in habitat that would otherwise have been suitable for breeding.

Research on amphibians also found evidence that reproductive output may be diminished in frogs breeding near highways because of acoustic interference (Barass 1985 in Larkin et al. 1996).

4.12.4 Impacts of Recreational Activities on Wildlife and Plant Communities

The effects of recreational activities in wetlands do not appear to be very well studied given the lack of recent articles on this topic. Most of the available information is anecdotal and focused on the more evident impacts such as loss of vegetation from the use of off-road vehicles. There is less information on the effects on wildlife of such disturbances as noise, light, glare, and human presence caused by recreational activities, particularly with respect to wetlands. None of the studies described in this section were located in the Pacific Northwest.

A synthesis paper on management of amphibians in Montana notes that among the many factors that are likely to contribute to a decline in amphibian populations are trail development, on- and off-road vehicle use, and development and management of recreational facilities (Maxell 2000). Citing a number of studies from the 1980s, Klein (1993) notes that recreational uses in natural areas can disrupt:

- Wildlife foraging and social behavior
- Animals that are feeding
- Parent-offspring bonds
- Pair bonds

The author also cites several studies stating that increased predation of nests and decreased densities of wildlife result from greater human recreational use of natural areas.

A study of flooded gravel pits in Britain examined the abundance and distribution of one species of wintering waterfowl with regard to recreational disturbance (Fox et al. 1994). The authors found that water-based recreational activity, such as boating, reduced the number of birds on the ponds to the greatest extent of all the observed activities. Ponds where fishing, walking, or other bank-side activities were allowed also showed reduced numbers of birds in comparison to the ponds that were designated reserves with restricted access. They were, however, not as reduced in abundance as those ponds that also allowed water-based activities.

The effects of recreational use on waterfowl were also studied in a nearshore area on Lake Erie (Knapton et al. 2000). Excessive human disturbance reduced the foraging efficiency and body fat acquisition for waterfowl and can result in decreased bird densities. Diving ducks appeared to be the most sensitive to disturbance. Furthermore, recreational shooting poses additional threats to wildlife if lead shot is used. The impacts of lead shot are discussed in Section 4.8.8.

In another study on recreation impacts on birds, Klein (1993) studied the specific behaviors of humans that disturb wildlife on a subtropical barrier island that is a National Wildlife Refuge off the coast of Florida. Her study sites were primarily in mudflat and mangrove wetland habitats. She tested a variety of treatments such as driving by without stopping, stopping the vehicle with and without getting out, approaching the birds on foot, and playing noise tapes. The author found that most of the bird species present were disturbed by the noise tape. Some species such as great blue heron consistently flew away when approached by a person, whereas other species tolerated human presence until closely approached.

Klein (1993) concludes that car traffic is less disruptive to wildlife than out-of-vehicle activity. Frequent human approaches may cause some bird species to forage in areas with fewer intrusions. Wildlife photographers were the most likely visitors to approach birds. Visitors who spoke with refuge staff and volunteers were the least likely to disturb birds, possibly due to an increased awareness of the needs of wildlife. While this study involved a very different ecosystem, it is useful because it generated data on bird species that also occur in Washington. It also is one of the few studies that examined the effects of specific human behaviors on wildlife.

Recreational activity is believed to be one of the main factors in lakeshore deterioration and decline in reed-dominated wetlands in a study of Central European lakes (Ostendorp et al. 1995). It is likely that trampling of bank-side vegetation by recreationalists is causing bank erosion and excessive siltation in nearshore wetlands.

Although recreation often occurs in more rural habitats, urbanization also brings increased intensity of recreational uses within remaining greenbelts and open spaces. A study in western Australia examined the trend in smaller lot size relative to the owners' use of nearby open spaces (Syme et al. 2001). Smaller lot size resulted in an increase in recreational visits by the homeowners to nearby wetlands. Increased access to and recreational use of wetlands is clearly one of the impacts that accompany urban development.

4.12.5 Impacts of Invasion by Exotic Species

Urban, suburban, and agricultural developments increase the likelihood of introducing exotic animal and plant species to wetlands. The following factors have been found to increase the opportunity for introducing exotic species:

- Increased access to wetlands through higher road densities
- Greater fragmentation of the landscape
- Higher densities of human land use
- Alterations of wetland hydroperiods
- Direct disturbance of wetlands

Exotic wildlife that has been introduced to the Pacific Northwest affects wetlands and wetland wildlife. The studies cited in the following discussion implicate disease, predation, and competition as major factors in limiting the success of native wildlife. While some causal relationships are clear, such as starlings displacing cavity-nesting ducks, others are less understood.

4.12.5.1 Impacts of Exotic and Invasive Plants in Wetlands

Invasive plants, especially non-native invaders, significantly alter the composition of plant communities in many wetlands, sometimes even forming nearly monotypic stands (Adamus et al. 2001). Changes in the plant community can be expected to result in changes to all the invertebrates and microscopic organisms that are associated with specific plant species.

Among the most geographically widespread invaders in Washington's wetlands are reed canarygrass (*Phalaris arundinaceae*), purple loosestrife (*Lythrum salicaria*), giant reed (*Phragmites* sp.), and European milfoil (*Myriophyllum spicatum*). Their increased dominance is frequently considered to be a result of human disturbances such as the following:

- Changes in soils or water levels within a wetland and/or the surrounding landscape, including accelerated sedimentation, eutrophication, and the construction of mitigation wetlands (Confer and Niering 1992, Magee et al. 1999)
- Changes in hydroperiod following urbanization (Cooke and Azous 2001)
- Increased human access and mechanical disturbance of wetlands (e.g., a study in southern Australia showed that vegetation removal and site disturbance are major factors in plant invasions; Detenbeck et al. 1999)

4.12.5.2 Impacts of Domestic Pets

Residential development typically brings increased access to wetlands by domestic pets, primarily cats and dogs. A study of house cat predation in Australia indicates that small mammals were the preferred prey of house cats, but cats also killed birds, reptiles, and amphibians (Barratt 1997). Many of the mice and rats collected by the cats in this study are exotic species themselves, but the results suggest that house cats may have significant impacts on native populations as well, particularly along the fringes of suburban expansion where native mammals are more common.

A similar study of house cat predation in Virginia determined that individual cats caught an average of 26 prey in urban areas and 83 prey in rural areas over an 11-month period (Mitchell and Beck 1992). Extrapolating these numbers of prey individuals to the total number of cats in a specific urban or suburban area would give an astonishingly high prey toll related to house cats. Taxonomic groups that were preyed upon in this study included birds, mammals, and reptiles.

4.12.5.3 Impacts of Exotic Wildlife

In Washington and Oregon about 42 exotic vertebrate species have established populations (Witmer and Lewis 2001). These include species of 18 birds, 19 mammals, three reptiles, and two amphibians. The birds were mainly introduced for hunting or aesthetic purposes, while the mammals mostly escaped from commercial or domestic settings. The amphibians and reptiles were released pets or were introduced for food or aesthetic purposes. About 30% of these species are restricted to freshwater and riparian systems, although others among this group will commonly use these habitats.

Some of the ecological consequences of these introductions for wetlands and wetland wildlife are well documented. Many introduced birds are known to usurp nests of native birds or to compete with them for nest sites. European starlings (*Sturnus vulgaris*) are known to displace wood ducks, woodpeckers, and other species from their nests, often destroying the eggs and young (Witmer and Lewis 2001). Starlings also out-compete many native species for nest cavities, overwhelming them with their large numbers and aggressive behavior. Transmission of disease, particularly from exotic birds and Old World rodents, is also a major problem that threatens native wildlife (Witmer and Lewis 2001).

Introduced mammals affect native wildlife and plants through predation and herbivory (Witmer and Lewis 2001). For example, nutria (*Myocaster coypus*), which were introduced from South America for fur production, have tremendous impacts on wetland vegetation, uprooting plants as they dig for rhizomes and denuding vast areas (Mitsch and Gosselink 2000, Witmer and Lewis 2001). Nutria may be implicated in population declines of muskrats (*Ondatra zibethicus*), probably due to competitive exclusion (Witmer and Lewis 2001).

4.12.5.4 Impacts of Exotic Invertebrates in Wetlands

Humans have introduced a number of non-native invertebrates to wetlands. Native invertebrate communities seem ill-adapted to compete with or avoid these alien species, but data on long-term effects to wetland communities are mostly lacking.

The zebra mussel (*Dreissena polymorpha*) has invaded many aquatic systems throughout North America (d'Itri 1997). This species can totally carpet substrates, displacing native mussels (Tucker and Atwood 1995), as well as some midges, snails, and caddisflies. The mussel has minimal or positive effects on amphipods and flatworms (Wisenden and Bailey 1995). They may also concentrate contaminants, making them more available to invertebrate food chains (Bruner et al. 1994). The rapid spread of zebra mussels may have been made more possible by the preceding decline of native mussels as a result of pollution and changes in habitat (Roberts 1990, Nalepa and Schloesser 1993, Hebert et al. 1991, Mackie 1991, Haag et al. 1993, Whittier et al. 1995).

Because unionid mussels in rivers are relatively immobile and have long life spans (often over 10 years), they are particularly susceptible to disruptions from introduced mussels as well as from impoundments and channelization (Mehlhop and Vaughn 1994). Riverine wetlands with higher alkalinity tend to be more susceptible to invasions by zebra mussels (Whittier et al. 1995). Wetlands along rivers might serve as refuges for native mussels otherwise impacted by expansion of the zebra mussel population (Tucker and Atwood 1995).

4.12.5.5 Impacts of Exotic Amphibians in Wetlands

The effects of exotic species of amphibians on native amphibians that use wetlands are particularly well studied. Predation and competition from introduced amphibians has been suggested as one cause of population declines for native amphibian species (Witmer and Lewis 2001).

Bullfrogs (*Rana catesbeiana*) are often cited as a factor in declining amphibian populations (Hecnar and M'Closkey 1996, Kiesecker and Blaustein 1998, Adams 1999).

Native to eastern North America, bullfrogs were introduced to the Pacific Northwest in the early 1900s for hunting and food. The species establishes easily in the wild because it can colonize a variety of aquatic habitats and is a prolific breeder. Bullfrogs are suspected of causing amphibian declines because they prey on frogs and salamanders and are often so numerous in wetlands that they are thought to out-compete native species for space (Witmer and Lewis 2001).

Studies of the role that bullfrogs play in declines of amphibian populations are, however, somewhat contradictory in their findings (Hecnar and M'Closkey 1996, Kiesecker and Blaustein 1998, Adams 1999, Witmer and Lewis 2001). It is possible that the effects of bullfrogs may differ for various species, or their influence may be quite subtle and complex.

Furthermore, several studies conducted in the Pacific Northwest have found either weak or no correlation between bullfrog presence and amphibian richness and abundance (Adams 1999, Richter and Ostergaard 1999, Richter and Azous 1995). Data from a monitoring program of amphibians in King County wetlands showed that bullfrogs are not causing competitive exclusion of native species (Richter and Ostergaard 1999). Native amphibian richness was not negatively correlated with bullfrog presence or with the presence of permanent water in the wetlands (Richter and Ostergaard 1999). Richter and Azous (1995) noted relatively high species richness for native amphibians in permanently ponded wetlands, the preferred habitat for bullfrogs.

Focusing on red-legged frogs (*Rana aurora*) in Puget Lowland wetlands, Adams (1999) concluded that this species is not excluded from wetlands that also support bullfrogs. The study showed little to no negative correlation between red-legged frogs and bullfrogs. It noted that exotic fishes such as sunfish, yellow perch (*Perca flavescens*), and smallmouth bass (*Micropterus dolomieui*) had greater effect on amphibian richness in the wetlands studied.

A study of red-legged frogs in the Willamette Valley in Oregon, however, stated their development was affected by both bullfrogs and exotic fishes (Kiesecker and Blaustein 1998). In this study, tadpoles showed decreased mass at metamorphosis and increased time to metamorphosis in the presence of larval and adult bullfrogs. Smallmouth bass alone had little effect on tadpole development, but red-legged frog tadpoles altered their use of microhabitats when both bullfrogs and smallmouth bass were present. Survival of tadpoles was affected only when both bullfrog adults and larvae were present, or when both bullfrog larvae and smallmouth bass were present.

Leonard et al. (1993) surveyed populations of the northern leopard frog (*Rana pipiens*) in Washington State. They found that the species had been extirpated from most of its historic range, with only small populations remaining in parts of eastern Washington. These authors noted that areas once inhabited by the northern leopard frog support exotic species, including bullfrog and such fish species as largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), yellow perch, and brown bullhead. They theorized that these species may be implicated in the decline of the northern leopard frog but have no definitive data to support this hypothesis.

In a review of studies across the country on potential causes of frog population declines, Hayes and Jennings (1986) concluded that existing studies do not support the theory that bullfrogs are a major cause. They argue that predation by exotic fishes may be a more likely hypothesis but note there is little data to support this. However, studies in the Pacific Northwest appear to support this theory (Adams 1999, Aker 1998). A study in the Okanogan Highlands in northeast Washington showed that richness of pond-breeding amphibian and abundance were diminished by the presence of exotic fish (Aker 1998). The non-native fish species observed in this study included largemouth bass, tench (*Tinca tinca*), brook trout (*Salvelinus fontinalis*), and perch. While there was lower amphibian richness in ponds with native fish than those with no fish, the data indicate that non-native fish had a greater impact on amphibian numbers and richness.

4.12.5.6 Impacts of Exotic Fish in Wetlands

Non-native fish have been widely introduced into waters of the United States, both intentionally and by accident. Adamus et al. (2001) cite research showing that the effects of invading species on native fish communities are usually adverse (Baltz and Moyle 1993), especially when coupled with simultaneous impacts from other factors (Larimore and Bayley 1996, Marschall and Crowder 1996).

The introduction of carp has resulted in significant impacts on wetlands in eastern Washington. Large, herbivorous fish such as carp compete directly with birds for submerged aquatic plants (Bouffard and Hanson 1997). The fish also resuspend the sediments on the bottom of lakes and ponds, and this has a significant impact on invertebrates as well as the submerged aquatic plants (see Section 4.5.5).

4.12.6 Summary of Key Points

- Alteration of soils can change the plant community in a wetland and allow invasion by exotic species.
- Noise creates stress for wildlife, but the impacts are very specific to individual species and to the type of noise generated.
- Recreational use of wetlands impacts the normal behavior of wildlife and reduces densities.
- Invasions by exotic species can alter the distributions of both plant and animal species in wetlands. The impacts of bullfrogs on other amphibians, however, are ambiguous even though this question has been studied extensively.

4.13 Chapter Summary and Conclusions

Humans create many different types of disturbances that can affect the environmental factors that control the performance of wetland functions. These disturbances were reviewed in Chapter 3. Chapter 4 has reviewed the information available on how these human disturbances impact wetlands and their functions. The disturbances that impact wetlands the most include:

- Direct changes to the physical structure of wetlands via filling, vegetation removal, tilling of soils, and compaction of soils
- Changes in the amount of water in wetlands
- Changes in how water levels fluctuate (frequency, amplitude, direction of flows)
- Changes in the amount of sediment
- Increases in the amount of nutrients
- Increases in the amount of toxic contaminants
- Changes in the amount of acidity
- Increasing the concentration of salts
- Decreasing the connection between habitats
- Other disturbances that are not as well documented including alteration of soils, construction of roads, noise, recreational access, and invasion of exotic species

Table 4-3 reviews how various land use practices create disturbances that can change the environmental factors that control wetland functions. Table 4-4 summarizes the effects of each of these disturbances in terms of the wetland functions they may impact. The rating of the impacts in the table represents a synthesis by the authors of all the information presented in this chapter. By combining the information in these two tables, it is possible to associate changes in functions of wetlands with general types of human land use, as shown in Table 4-5.

For example, Table 4-3 shows that urbanization creates significant disturbances that change the amount of water, the fluctuations of water levels, and input of sediments, nutrients, and contaminants to wetlands. Table 4-4 shows that disturbances to water flows, fluctuations of water levels, and input of sediments, nutrients, and contaminants have a significant impact on the wetland functions of providing habitat for plants, invertebrates and reptiles/amphibians. Table 4-5 synthesizes this information to show that urbanization impacts the habitat for plants, invertebrates, reptiles, and amphibians in wetlands. The human land uses create various disturbances in the environment, and those disturbances in turn affect the factors that control wetland functions, ultimately leading to changes in those functions.

The scientific information available indicates that human activities and uses of the land can have significant impacts on the functions in wetlands at both the larger, landscape scale and at the scale of the individual wetland itself. As a result many different approaches and methods have been developed to try to minimize these impacts. These methods include regulations to control human activities near wetlands, methods to replace the functions, and ways to protect the wetland resource through restoration. The effectiveness of some these tools at actually protecting wetland functions are discussed in Chapters 5 and 6.

Table 4-3. Disturbances resulting from different land use practices that can change the factors that control wetland functions.

Key to symbols used in table:

(xx) land use creates a significant disturbance of environmental factors

(x) land use creates a disturbance

(nm) studies on impacts of this land use do not mention this disturbance

(h) literature is lacking but disturbances can be hypothesized based on authors' experience

(?) information lacking

Disturbance	Scale of Disturbance	Agriculture	Urbanization	Mining
Changing the physical structure within wetlands (filling, vegetation removal, tilling of soils, compaction of soils)	Site scale	xx	xx	h
Changing the amounts of water	Landscape scale Site scale	xx xx	xx xx	? h
Changing fluctuations of water levels (frequency, amplitude, direction of flows)	Landscape scale Site scale	xx xx	xx xx	? h
Changing the amounts of sediment	Landscape scale Site scale	xx xx	xx xx	h h
Increasing the amount of nutrients	Landscape scale Site scale	xx xx	xx xx	nm nm
Increasing the amount of toxic contaminants	Landscape scale Site scale	xx xx	xx xx	x xx
Changing the acidity	Landscape scale Site scale	nm nm	nm nm	x xx
Increasing the concentrations of salt	Landscape scale Site scale	x x	nm nm	nm nm
Decreasing the connection between habitats	Landscape scale	xx	xx	h
Other disturbances	Site scale	xx	++	h

Table 4-4. Synthesis of the information reported in the literature on the impact of different human disturbances on wetland functions.

Key to symbols used in table:

++ Significant impacts on specific functions have been documented
+ Some data suggest impacts or impacts could be hypothesized
0 Data indicate that impacts are minimal
? Information is lacking

	Functions							
Disturbance Type	**Hydrologic**	**Water Quality**	**Plants**	**Habitat for Invertebrates**	**Habitat for Amphibians and Reptiles**	**Habitat for Fish**	**Habitat for Birds**	**Habitat for Mammals**
Changing the to physical structure of wetland	+	+	++	++	+	+	++	+
Changing the amount of water	+	+	++	++	++	+	+	?
Changing fluctuations of water levels	?	?	++	+	++	+	?	?
Changing amounts of sediment	+	?	++	++	?	?	?	?
Increasing amounts of nutrients	+	+	++	++	++	+	+	+
Increasing amounts of toxic contaminants	?	+	++	++	++	++	++	?
Changing acidity	0	+	+	++	++	+	+	+
Increasing concentrations of salt	0	?	++	++	?	?	+	?
Decreasing connections between habitats	0	?	?	?	++	?	++	+
Other disturbances	?	?	++	+	++	++	++	++

Note: A (++) does not indicate the direction of the impacts to functions. In some cases the disturbance can increase the function or the richness and abundance of species and in other cases it can decrease them. A disturbance can also decrease or increase a function depending on the intensity of the disturbance (e.g., small amounts of nutrients can increase invertebrate richness and abundance, but too much will cause eutrophication).

Table 4-5. Synthesis of the impacts of different land uses on wetland functions.

Key to symbols used in table:

++ Significant impacts on specific functions have been documented
+ Some data suggest impacts or impacts could be hypothesized
? Information is lacking
+? Some impacts have been documented but more information is needed

Land Use	**Functions**							
	Hydrologic	**Water Quality Improvement**	**Plants**	**Habitat for Invertebrates**	**Habitat for Reptiles and Amphibians**	**Habitat for Fish**	**Habitat for Birds**	**Habitat for Mammals**
Agriculture	+	+	++	++	++	++	++	+?
Urbanization	+	+	++	++	++	++	++	+?
Mining	?	?	+	++	++	+	+	+?

Note: A (++) does not indicate the direction of the impacts to functions. In some cases the land use can increase the function or the richness and abundance of species and in other cases it can decrease them. A land use can also decrease or increase a function depending on the intensity of the land use.